Measuring Drinking Water Affordability and Sustainability

by

Jessica J. Goddard

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Committee in charge:

Professor Isha Ray, Co-chair
Professor Rachel Morello-Frosch, Co-chair
Professor John Radke
Dr. Carolina Balazs

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Abstract

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Access to safe water is necessary for life, but safe water is not always affordable or accessible. The human right to water and the Sustainable Development Goals are landmark social visions for a world where everyone has access to safe and affordable water. These frameworks embed water access in broader aspirations for sustainably managing resources on earth. Metrics play a key role in developing policies and analyzing progress toward water access equity and sustainability. This dissertation contributes to scholarship on water access, economics, and sustainability through developing metrics of household water affordability and greenhouse gas impacts of bottled water.

The impacts of unaffordable water can be substantial for economically vulnerable households, who may compromise health and food-related expenses to pay for water and utilities. In a comprehensive and critical review in Chapter 2, I evaluate the state of water affordability research to understand how water affordability should be measured to advance the human right to water and Sustainable Development Goals. In the following chapter, I aim to operationalize metrics for household water affordability in California as part of the state's first human right to water tracking tool. The research in Chapter 3 offers insight into California's water affordability challenges through the development of three water affordability indicators, as well as analyses by system size and poverty levels. At the same time, this study underscores the substantial data gaps facing researchers and policymakers aiming to improve water access equity in the U.S.

In Chapter 4, I turn to the question of water access sustainability in Mexico, where water access is high, but trust in water quality is low–resulting in the highest per capita consumption of bottled water in the world. I develop a representative metric of greenhouse gas emissions associated with household bottled water use in Mexico using life cycle assessment modeling. To realize sustainable transitions, research into the antagonistic or synergistic interactions among Sustainable Development Goals is required. Achieving water access through the use of bottled water is a prime case study to evaluate such interactions and identify areas for emissions reductions on the pathway to water access for all.

The projects in this dissertation enable me to quantify equity and sustainability dimensions of household water access in novel ways, and I explore the scholarship and policy implications of this work in a concluding chapter.

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INTRODUCTION

As of 2019, 2.1 billion people did not have access to safely managed drinking water and of these, 785 million people lacked basic drinking water service worldwide (United Nations 2019b). Safe drinking water is essential to our ability to thrive (Villanueva *et al* 2014). Lack of access is a risk factor for people's health and may result in significant psychosocial stress (Bisung and Elliott 2017). Households without water access are forced to develop coping strategies (Nastiti *et al* 2017a, Pattanayak *et al* 2005)–involving walking long distances (Pickering and Davis 2012), timing their day around access to intermittent borewell access (Burt *et al* 2018), and relying on a patchwork of different sources (Smiley 2016). For many years, coping with poor water quality was not widely recognized as a reality in rich countries like the U.S. (Wescoat *et al* 2007). Though a majority of households in the U.S. have access to a piped water supply, water quality is not always safe (Evans *et al* 2019, Allaire *et al* 2018) or affordable (Teodoro 2019).

In the U.S., by most measures, water access problems are worsening. The large-scale water infrastructure that radicalized safe drinking water access in the early 20th century U.S. is now beyond its design lifetime, leading to leaking pipes and worse water quality at the tap (National Academy of Sciences 2019). Many factors influence the safety and affordability of water, including water system governance (McFarlane and Harris 2018, Berg 2016, Bakker 2010), household and water infrastructure quality, and source water quality (Price and Heberling 2018). Additionally, the water infrastructure of today was not built to be resilient against a changing climate. Drought, wildfires, sea-level rise and salt water intrusion are increasing in intensity, threatening the reliability, quality, and sustainability of society's drinking water systems (Garrick and Hall 2014).

The human right to water and the Sustainable Development Goals (SDGs) are landmark social visions for a world where everyone has access to safe and affordable water (United Nations Committee on Economic Social and Cultural Rights 2002, United Nations 2015). The UN framework defines safe, affordable, and accessible water and sanitation a fundamental human right (United Nations Committee on Economic Social and Cultural Rights 2002). The Sustainable Development Goals use embed this definition of the human right to water in a broader set of sustainability targets for all member nations of the UN by 2030 (United Nations 2015). While access to improved (i.e. 'covered') drinking water sources has increased 10% since 2000 (United Nations 2019b), a primary critique of the human right to water and SDGs is that they are aspirational but lack teeth. My perspective is that the human right to water and SDGs are a critical part of the work required to develop alternative social-ecological imaginaries about the world we want to live in. Both frameworks respond to the world's unsustainable use of land, energy, and water and the inequitable distribution of these resources.

How might these aspirational visions and framings for sustainable water access work *for* the communities that are disproportionately paying the socioeconomic and health costs of inequitable and degrading water systems? At a practical level, the human right to water and the SDGs support increased public data collection, monitoring, and benchmarking. Communities have also used these broader frameworks to initiate local action on improving water access and to align environmental justice goals with the human right to water (Harris *et al* 2015, Sultana and Loftus 2015). As with other human rights, indicators and metrics play a role in holding governments accountable in international human rights law, structuring reports and investigations into the progressive realization of rights, and monitoring trends over time (Meier *et al* 2017). Entire research agendas are now framed by the SDGs, marking the mobilization of research funds to study pathways for sustainable transformations (Sachs *et al* 2019).

Critically, the human right to water and SDGs are not solutions or even blueprints. For example, nothing about the human right to water or the SDGs precludes profit-seeking and (undemocratic) private ownership of essential resources. However, in addition to safe and affordable water access for all, the SDGs broadly call for decoupling of consumption from fossil fuels. As such, the SDGs confront a fundamental driver of our current economic system, putting into question the relationship between the economics that drive global markets and the resources essential for life. Social equity and sustainability are necessarily joint goals in the SDG paradigm, but operationalizing this is not a trivial effort.

DISSERTATION OVERVIEW

It is at the intersection between the human right to water, the SDGs, metric development, and the science-policy interface that I situate the research in this dissertation. Each of my chapters reflects deeply collaborative work with scientists and co-authors that work in environmental justice, environmental engineering, sustainability science, and water policy to improve water access for all. By investigating water access in California and Mexico, my goal is to develop analyses and tools that are useful in their social, political, economic, and geographic contexts, but that also provide insight into what it means to operationalize the right to water. I use the human right to water and SDGs as guiding frameworks with clear normative aspirations for metric and indicator development.

Ultimately, I aim to develop quantifiable metrics to support policies and monitoring for sustainable, safe and affordable domestic water. The chapters that follow operationalize metrics to capture two aspects of water access that have eluded adequate monitoring and development: water affordability and the sustainability of household drinking water choices. By comprehensively reviewing metrics for water affordability (Chapter 2), developing new metrics and empirically investigating them in the case of California (Chapter 3), my work advances the field of water affordability for human right to water frameworks. In Chapter 4, I turn to the question of bottled water sustainability as a household water source in low and middle income countries, using Mexico as a case study. These contexts enable me to quantify equity and sustainability dimensions of household water access in novel ways with direct application to policy solutions.

My methodological home is at the intersection between environmental social-sciences and sustainability science. This work is problem driven, and thus necessarily interdisciplinary. The three core chapters are broadly housed within the human right to water and SDG paradigms, drawing on theories and tools from water access scholarship (water governance/economics), environmental justice, and environmental engineering. I use critical literature reviews, surveys and participant-observation through a co-production model of research, data analysis and statistics, and life cycle assessment.

In Chapter 1, I introduce key concepts and theories from critical geography and economic philosophy as relates to the human right to water and Sustainable Development Goal frameworks. I attend to the history and critique of these frameworks. I conclude with a brief contextualization of my dissertation chapter case studies developing metrics in California and Mexico.

In Chapter 2, I conduct a comprehensive review of over 100+ publications spanning the fields of economics, water governance, and environmental justice scholarship. I use human right to water and SDG concepts of affordability, equity, and sustainability to critically evaluate existing metrics. Distilling critical themes on how and what to measure, I offer a conceptual framing and best practices for measuring water affordability.

In Chapter 3, I develop an approach for evaluating water affordability in California's first human right to water tracking tool in co-production with staff scientists at California's Office of Environmental Health Hazard Assessment (OEHHA). Few studies investigate water affordability in California, but affordability is a pressing issue (Christian-Smith *et al* 2013, Pierce and McCann 2015). Following national trends, water bills in California have been increasing two to three times faster than inflation in urban areas whilst average incomes remain stagnant nationwide (Hanak *et al* 2014). This suggests that affordability problems are on the rise–and in this chapter I develop metrics to measure this phenomena in community water system across the state. I then analyze the results disaggregated by a key characteristic of system vulnerability–water system size. In this work, I am committed to representing as many water systems for which data is available. Previous studies have limited their research to large water systems or case studies in small systems. In trying to capture the state of all community water systems, it became clear that that many water systems do not report water bills or have adequate data. To mitigate bias, I analyze predictors of missing data and incorporate potential confounders of missingness in a model of affordability ratios. We assess reliability for secondary data sources from the census and state surveys, including a survey 60+ water systems. The lack of information access for policy-makers trying to monitor the human right to water underscores substantial gaps in monitoring the human right to water when there are thousands of water systems and limited resources.

In Chapter 4, I investigate sustainability in household water access. I design a processed-based life cycle assessment for the energy requirements of bottled water in Mexico (known as garrafones) over seven stages of production and distribution. Though data on bottled water use patterns are scarce, bottle reuse is high and the market share of micro-to small-scale purification plants is large across Mexico. I draw on expert interviews and multiple data sources to parameterize the model and develop several scenario analyses to estimate greenhouse gas emissions. With bottled water use rising quickly in emerging economies, Mexico as a case can inform the potential sustainability implications of pursuing water access through bottled water in other countries.

Taken as a whole, I hope this work both informs California's realization of the human right to water, on the one hand, and contributes to the way we evaluate household water access to include ecological sustainability, more generally in our pursuit of the SDGs.

CHAPTER 1

1. THE HUMAN RIGHT TO WATER AND SUSTAINABLE DEVELOPMENT GOALS

This chapter first outlines key debates on the right to water and how the right to water relates to sustainable access through the Sustainable Development Goals. From this global and theoretical view, I turn to the local contexts of California and Mexico, introducing how these cases reflect different instantiations of the right to water and sustainable water access. It is essential to begin the story of the human right to water in its international context, for it contextualizes my case studies in the broader dialogue on water access. This serves as an overview of broad theories and frameworks relevant to the remainder of the dissertation.

1.1 GLOBAL BEGINNINGS

At the turn of the century, around 1.1 billion people globally still lacked adequate access to clean water and over double that number lacked access to sanitation (WHO/UNICEF 2019). This was the state of things despite a century of large-scale, public investment and ownership in water and sanitation supply worldwide. The public model of ownership had failed to connect *all* people to piped networks by the end of the 1980s, and this partially legitimized the global transfer of water ownership and management from the public to private sector, which promised to remedy these 'inefficient' and inadequate publicly-owned systems. In the Global South, this was accelerated and promoted through loans from development banks like the World Bank and the International Finance Corporation. But private sector involvement advanced in the UK and U.S. as well (Bakker 2001). As we know today, the many constellations of private sector involvement in water and sanitation *also* failed people globally, without necessarily advancing gains of efficiency and environmental performance, as promised (Davis 2005, Mirumachi 2011). Numerous case studies in the Global South illustrate the complex and often painful failures of public, private, or hybrid public-private partnerships to adequately provide safe and affordable water (Smith 2012, von Schnitzler 2008, Kooy and Bakker 2008, Loftus 2009). The human right to water emerged as a political movement largely in response to the failures of global water privatization (Bakker 2010).

The human right to water rests on claims that water is not substitutable, that water is embedded in other rights, (such as food) and that legal protection of water access for all might limit private sector involvement. Distinct from 18th century concepts of rights like freedom, the right to water is considered a "second generation" right, also known as a welfare or subsistence right. International documents on the right to water proliferated in the 1990s (Gawel and Bretschneider 2016), but the United Nations General Comment 15 (GC15) remains the flagship document on the matter (United Nations Committee on Economic Social and Cultural Rights 2002). GC15 states that sufficient amounts of water should be safe, accessible, affordable, and acceptable for domestic use and sanitation. In 2010, 122 countries (not including the U.S.) signed United Nations General Assembly Resolution A/Res/64/292. The resolution obligates ratifying states to “respect, protect, and fulfill” the rights to water and sanitation as outlined in GC15 (UN General Assembly 2010).

The 2010 Resolution shifted the language from “the right to water and sanitation” to“the rights to water and sanitation.” By using “rights” in the plural, the document aimed to clarify that the right to water and the right to sanitation are separate but equally important rights.

In this dissertation, I use the terms water, drinking water, and domestic water interchangeably to represent water used for consumption (drinking, food), hygiene (showering, washing hands), and minimal sanitation needs (i.e. water to flush the toilet). This extends the definition of domestic water used by Howard and Bartram 2003, who define domestic use as consumption, hygiene, amenities, and productive uses, but they do not clarifying the use of household water for toilet flushing, which is often implicit in water bills (Howard and Bartram 2003). Amenities (lawn watering) and productive uses (e.g. water for livestock) are other important household water uses, but these are not the focus of this dissertation. I focus my analysis on domestic water use in Chapters 2 and 3, and on drinking water for consumption in Chapter 4. The approaches I develop can be generalized to include sanitation/wastewater, but the analytical focus of this work is domestic water use.

1.2 Human Right to Water in the Sustainable Development Goals

The United Nations Environment Programme was developed in the 1970s and has been an institutional umbrella for global efforts to reduce poverty and increase sustainability for 50 years. The Sustainable Development Goals–announced in 2015 as a 15 year agenda–reflect a vision rich with ambition across 17 main goals (Figure 1.1). SDG 6.1 aims to “achieve universal and equitable access to safe and affordable drinking water for all” benchmarked by the indicator “Proportion of population using safely managed drinking water services” (United Nations 2019a). This is widely recognized as language building on the GC15 and other efforts for the human right to water. Indeed, the right to water provides the philosophical underpinnings for water access as an SDG. SDG 6.1 is also recognized as supporting other SDGs like poverty alleviation (SDG 1) and good health and wellbeing (SDG 3) (Pradhan *et al* 2017).

SDG 6 extends the traditional right to water framework by including several sustainability-related aspirations and targets. Sustainability dimensions of water access are discussed in targets on water use efficiency (Target 6.4), pollution minimization (Target 6.3), and source water protection (Target 6.6). Each of these targets are critical for ensuring the longevity and quantity of freshwater supply for human and ecosystem use. Similarly, action on all of these targets helps minimize energy consumption used in the distribution and treatment of water for households (and other water uses, like agriculture). As I discuss below, the sustainability of household water choices is not currently a focus of metrics for water access. In Chapter 4, I argue that this is problematic in areas where bottled water is the primary water source. By extending the human right to water to be part of a broader sustainability framework, the SDGs enable us to identify where pathways to water access may undermine broader sustainability aims.

Figure 1.1. United Nations Sustainable Development Goals (SDGs).

Shown are the 17 SDGs. Each goal has several targets and associated indicators.

1.3 KEY DEBATES ON THE RIGHT TO WATER

Below I summarize three critiques about the right to water, which are broadly categorized as commodity vs. rights, the institutionalization critique, and the feasibility critique. These critiques situate my analytical chapters in a broader conversation about the role of human rights and metrics of water access in realizing equitable outcomes.

1.3.1 *WATER AS A COMMODITY OR A RIGHT?*

The human right to water emerged as a counter-privatization campaign. Yet the Dublin Principles established in Rio in 1992 suggest that water can be both a right and a commodity: "Water has an economic value in all its competing uses and should be recognized as an economic good" and "it is vital to recognize first the basic right of all human beings to have access to clean water and sanitation at an affordable price" (emphasis mine, ICWE 1992). This statement is important because it highlights two dominant perspectives on water that persist today: water is both an economic good, and a human right. It is precisely this dual manifestation of water–as a good and a right–that leads critical geographer Karen Bakker to argue that, without attention to property rights, the human right to water is not an adequate response to the failures of water privatization (Bakker 2007).

In its fullest expression, privatization rests on the claim that water is a commodity (or economic good). Water as a commodity requires clearly defined private property rights, market exchange value, and commensurability with financial instruments (e.g.

bonds or derivatives) (Hahn *et al* 2015). The critical literature makes clear that water itself is not easily commodified (Bayliss 2014), but that the water sector (its infrastructure, ownership, and management) is increasingly financialized (Ahlers and Merme 2016). Water was first declared a human right in part to resist its commodification. By the early 2000s, however, corporations and purveyors of water privatization like the World Bank were also embracing the rights framework alongside grassroots activists (Bakker 2010).

This debate is important primarily because defining what water *is* determines how it is managed and valued. It also underpins what work a rights framing can and cannot do for improving water access. In human rights discourse, water is usually portrayed as a 'public good' (non-rivalrous and non-excludable). Characterized as a public good, water is defined by its non-substitutability, its requirement for life, and its necessity for public health. Relatedly, water has standing as a merit good – requiring public financing to support its consumption for the benefit of individuals regardless of their preference for the good. Bakker points out that conceptualizing water as a public good is somewhat of a mischaracterization, even if it has political significance and cultural meaning. Water as an economic good can indeed be rivalrous and potentially excludable (e.g. bottled water is a private good).

The Dublin Principles were making clear the mixed identity of water as a right and a good. Bakker's concept of water as an "uncooperative commodity" is most useful in communicating that water eludes traditional economic definitions (Bakker 2007). Bakker goes on to define water as "a common-pool resource, from which it is difficult to exclude access, but the consumption of which by one individual can reduce the benefits for others" (Bakker 2007). In this light, the commons–not human rights–is the adequate response to privatization. This argument is based in part on the fact that physical access to water has not guaranteed its safety (Bain *et al* 2014) or its affordability (Smets 2017) under public or private management and ownership (Bakker 2010).

Water can thus be a common pool resource, a merit good, a quasi-public good, and increasingly, a commodity. What does this mean for what a rights framing can do? On the one hand, the limitations of a human rights argument against the sanctity of private property rights in highly marketized economies should not be overlooked. Native American communities in California are still marginalized in the fight over property rights that ensure their access to water (Womble *et al* 2018). In cases where transfer of water to community ownership is unlikely, however, a rights framing may provide social goals, standards of service, and recourse for communities to ensure equitable water access and to limit the effects of privatization. In industrialized water systems (such as California), a transfer of water rights to the commons, or community ownership, is unlikely. Bakker recognizes that some areas are more likely to have community engagement and governance rather than ownership.

In these contexts, one value of the human right to water is to forward a vision with the clear end goal of equitable safe water access. This is especially useful in light of debates on private vs public provision, which often miss site of the end goal of water access for all (Mirosa and Harris 2012). For example, the human right to water concepts of inequality and social vulnerability has been leveraged as a basis for solidarity between local and global efforts (Sultana and Loftus 2015, Harris *et al* 2015). Another key role of the human right to water discourse has been to mobilize legal obligations and committ states and governments to ensuring access to water for all–regardless of the ownership or provision structure (Gupta *et al* 2010).

1.3.2 *Who is Going to Implement, and How?*

Amartya Sen classifies critiques of human rights (generally) as usually one of two types: the institutionalization critique or the feasibility critique (Sen 2004). Example questions that fall under the institutionalization and feasibility critiques are: a) how much water is protected by the human right to water?; b) who is accountable to the human right to water?; c) how should water be priced (free?, 'affordable', 'full cost'?); d) will the human right to water ignore and worsen environmental sustainability?; and e) how will the right to water interact with already-established water rights (Bakker 2010)? In other words – who will implement it, and how?

The institutionalization critique effectively argues that a right cannot exist if there is no obligation-bearer, i.e. an institution. The fact that an institution *could* be created to deliver a right is not sufficient to establish the right. Sen's counter to this critique is that secondary rights (like the right to water) are not only obligations on the institutions that can enable them. They are also imperfect obligations (in the Kantian sense), meaning that that there are multiple ways to fulfill them. In other words, there is no agreed upon or necessarily correct way to enable rights. Questions like: How much gets protected? What price? do not have "correct" answers or pathways for realization. This is especially true for normative aspects of the right to water, like affordability or accessibility which are largely questions about "reasonableness of burden" (Gawel and Bretschneider 2016) that require ongoing evaluation. This ambiguity is fundamental to the obligations, and thus cannot be a critique for dismissing the value of the rights framing. The concept of imperfect obligations also enables a wider view on what constitutes action towards fulfilling rights. For example, Sen emphasizes that social movements attempting to change institutions is an example of living out the obligation to subsistence rights even if the institutions themselves do not exist.

The feasibility critique underlies challenges to subsistence rights based on the idea that complete accomplishment of social and economic rights is nearly impossible from an implementation standpoint. Sen counters with the point that if rights are seen as unrealizable in the current moment, we need to focus efforts on changing the current circumstances that keep such rights unrealizable. This response offers a reframe of Bakker's argument against rights as a response to privatization–where we might see rights discourse as the catalyst to investigate deeper structural change in ownership while still maintaining a clear vision about the outcome (safe and affordable water access). In other words, if ensuring the right to water demands new ownership structures, then part of realizing the right to water may be through structural changes that bring down those barriers to access. Such requirements, however, do not negate the value of the rights framing nor do they weaken the role that the right to water has to play. A rights commitment does not, in short, detail the commitment to how the right is realized or its evaluation–but this is an insufficient ground to dismiss rights framings.

1.4 A BROADER VIEW OF WATER ACCESS

This dissertation takes the perspective that a human rights framework can be adaptive and provide a part of the broader solutions required to address water equity challenges. Bakker claims that the human right to water is "not the solution but [rather] a strategy for creating the context in which claims for social justice can be pursued" (Bakker 2010). Relatedly, Sultana & Loftus argue for activists and scholars to reclaim "the script from the technocrats" to enable the human right to water as "a political moment" (Sultana and Loftus 2015). Underlying this view is the perspective that rights frameworks have an "aspirational sense" and "progressive potential" (Sultana and Loftus 2015), which can serve as a counter-narrative to marketization and connect local water access narratives to global movements for the human right to water (Harris *et al* 2015).

Beyond providing an aspirational sense, Sen claims that human rights declarations are in effect ethical demands. Ethical demands relate to specific freedoms that rights privilege or enable. An ethical, rather than legal starting point allows Sen to articulate his now famous concepts of substantive opportunities—supported by a frame of capabilities–and freedom of processes (Sen 2005). Human rights are grounded in the content of ensuring substantive opportunities for individuals, i.e. ensuring the conditions for what a person would like to do and is able to do in a broad sense. This requires not only a focus on individual capabilities but also on the freedom of processes that enable different conditions (e.g. the availability of water and the institutions that enable this). Human rights frameworks can guide and direct equity and fairness in the processes that enable people to realize their rights.

There is a parallel between Sen's capabilities/process freedom approach in the context of human rights and Peluso and Ribot's 2003 articulation of "access" to resources such as land or water as:

> "the ability to benefit from things–including material objects, persons, institutions, and symbols. By focusing on *ability*, rather than *rights* as in property theory, this formulation brings attention to a wider range of social relationships that can constrain or enable people to benefit from resources without focusing on property relations alone" (2003).

Access analysis, in their view, is a study of the power that attends to the "means, processes, and relations by which actors are enabled to gain, control, and maintain access to resources" (Ribot and Peluso 2003). These mechanisms (means, processes, and relations) can be about property rights, but they can also be structural and relational mechanisms that define access. Examples include technology, politics, social relations and identities, access to capital, knowledge and authority. Access, or *ability* to benefit, is negotiated through these mechanisms.

1.4.1 *HUMAN RIGHT TO WATER AND ENVIRONMENTAL JUSTICE*

As noted above, human rights can provide aspiration, the context for claims and pursuit of social justice, and a shift in who owns the "script"–or discourse of water access. The work toward realizing the right to water may require new institutions and actions to realize the end goal of safe and affordable water access for all. Importantly, none of this is prescribed or predetermined. We thus need to look outside the global frameworks to identify plausible pathways, likely barriers, and contextual relevance.

Environmental scholarship is well poised to inform the institutionalization of SDGs and the human right to water by identifying barriers to access (or process freedoms, in the language of Sen) and directing benefits, or new mechanisms of access (in the Peluso and Ribot sense), for historically marginalized communities.

Environmental justice has grown into a transdisciplinary academic research field since 1982, when community members in Warren County, North Carolina protested years of unjust exposures to PCBs (Gelob 2005, Mohai *et al* 2009). The Warren County protests marked a nationwide moment in which the interrelated nature of civil rights and toxic environmental exposures made headlines. Early reports on environmental injustices pointed to unequal exposure to toxics along lines of race and income (e.g. the 1987 United Church of Christ study, *Toxic Wastes and Race in the United States*). Today, research continues to investigate exposure patterns and race (Schaider *et al* 2019, Brown 1995). Public health has extended environmental justice scholarship by deepening our understanding of disproportionate environmental impacts through the concepts of cumulative exposures (i.e. all environmental and social stressors over one's life course) (Solomon *et al* 2016, Sexton and Linder 2010) and differential vulnerability (DeFur *et al* 2007, Liu and Xu 2016). Neighborhood context (Diez Roux and Mair 2010), historical and regional segregation (Morello-Frosch and Lopez 2006), as well as social and racial inequality (Cushing *et al* 2015) have been identified as drivers of health disparities in the U.S.

Water access intersects with and can shape the social, physical-chemical, and economic burdens put on communities and households. Environmental justice findings can therefore provide insight into local implementations of the right to water and SDGs, which have normative aims to address historically marginalized groups, and to ensure equity and sustainability.

1.5 Drinking Water Access in the U.S.

My first two dissertation chapters focus on water affordability in the U.S. and California. In the U.S., the experience of unaffordable bills, poor water quality, or reliance on multiple–possibly unsustainable–sources of water is not equally distributed. Taking a rights and environmental justice view, scholarship has identified that disparities in drinking water access persist because of inequities in infrastructure (Vanderslice 2011), enforcement and regulatory design (Katner *et al* 2016, Cory and Taylor 2017, Cory and Rahman 2009), uneven monitoring (Heaney *et al* 2011), high water and bottled water costs paired with low ability-to-pay (Pannu 2012, Moore *et al* 2011), and low technical, managerial and financial capacity (Balazs and Ray 2014).

Water quality and equity have been studied across the U.S. in multiple case studies (Corlin *et al* 2016, Eggers *et al* 2018, Wedgworth and Brown 2013, Gibson and Pieper 2017, Stillo and Gibson 2017, Naman and Gibson 2015, Balazs *et al* 2012, 2011, Heaney *et al* 2013) and nationwide (Allaire *et al* 2018, McDonald and Jones 2018). Contaminant exposures have been demonstrated to disproportionately impact communities of color (Stillo and Gibson 2017, Balazs *et al* 2012) and marginalized groups (Jepson *et al* 2017). For example, non-White Hispanic populations are disproportionately exposed to nitrates in small water systems in the Central Valley (Balazs *et al* 2011), and a recent study found this positive correlation persists even outside of rural and agricultural communities across the U.S. (Schaider *et al* 2019). Households without citizenship had higher water insecurity than registered citizens in a case study in Texas (Jepson and Vandewalle 2016). Nationwide, drinking water quality violations were found to occur disproportionately in communities that are underinsured (McDonald and Jones 2018).

Water affordability, however, has only recently attracted the attention of scholars from multiple disciplines in the U.S. (Teodoro 2019, Pierce and McCann 2015) outside of utilities-focused publications (Beecher 1994). Balazs and Ray (2014) examined how unaffordable water compounds and perpetuates water quality problems, leading to a "joint burden" for vulnerable households and systems. If coping costs are too high, water systems and households may not be able to adequately cope with drinking water contamination. Yet even where households cannot afford water, they may still pay for it; water bills are often "paid but unaffordable" (Colton 2017). The impacts can be substantial for vulnerable households, who may compromise health and food-related expenses to pay for water and utilities (Cory and Taylor 2017) or turn to bottled water (Javidi and Pierce 2018).

1.5.1 *The Human Right to Water in California*

The water access challenges in the U.S. described above are, unfortunately, well represented in California. Yet the opportunities for addressing water access inequities are growing. California declared water a human right in 2012 with the monumental passage of Assembly Bill 685 (Eng 2012). This legislation–based on years of advocacy work by environmental justice groups across the state–has formally reshaped the priorities and objectives of water access research and policy in several key ways. Firstly, the human right to water codified a set of qualifiers for what is protected–that is, *safe, clean, affordable, and accessible* water (Figure 1.2). These adjectives have been ubiquitously adopted in policy and legal briefs (Salceda *et al* 2013, Environmental Law Clinic 2017), think tanks (Feinstein 2018), and among activists. They are now enshrined as part of California's Water Code (Eng 2012).

How does the right to water get defined, measured, protected, and realized in California? By what measures, at what scale, and for what aims do we need to analyze water as a human right? What interactions among the dimensions of water as clean, safe, affordable, and accessible exist, and how might this implicate our interpretation of water as a human right? Why, and for whom, is water unaffordable, unclean, and inaccessible?

In 2015, California's Environmental Protection Agency's (EPA) Office of Environmental Health Hazard Assessment (OEHHA) began the earliest work seeking to quantitatively assess the human right to water in California water systems (Balazs *et al* 2019). In late fall 2015, I joined Dr. Carolina Balazs and colleagues at OEHHA to work on developing the state's first human right to water tracking tool and report. The focus of my dissertation work has been the development of methods and measures to understand water affordability. Working at the science-policy interface through collaboration with OEHHA, the questions and metrics developed on water affordability in Chapters 2 and 3 have been the subject of academic scholarship, interagency review, and public comment through reports that build on the work herein. The public process and outreach dimensions of this work can be found at OEHHA's website: https://oehha.ca.gov/water/report/human-right-water-california.

In Chapter 2, I identify how several core dimensions relevant to water affordability as the right to water and as an SDG (Figure 1.3). I explore in depth the ways that scholars measure water costs (like a water bill or purchasing bottled water), and the impact this might have on households across the income distribution. I identify key debates on how much water is protected as a right, as well as the way that economically vulnerable groups get represented (or not) in measures for water affordability. Finally, the sustainability of provision–motivated by the SDGs–is incorporated in our representation of core dimensions relevant to household water affordability.

Assembly Bill No. 685

CHAPTER 524

An act to add Section 106.3 to the Water Code, relating to water.

[Approved by Governor September 25, 2012. Filed with Secretary of State September 25, 2012.]

LEGISLATIVE COUNSEL'S DIGEST

AB 685, Eng. State water policy.

Existing law establishes various state water policies, including the policy that the use of water for domestic purposes is the highest use of water.

This bill would declare that it is the established policy of the state that every human being has the right to safe, clean, affordable, and accessible water adequate for human consumption, cooking, and sanitary purposes. The bill would require all relevant state agencies, including the Department of Water Resources, the State Water Resources Control Board, and the State Department of Public Health, to consider this state policy when revising, adopting, or establishing policies, regulations, and grant criteria when those policies, regulations, and grant criteria are pertinent to the uses of water described above.

Vote: majority Appropriation: no Fiscal Committee: yes Local Program: no

THE PEOPLE OF THE STATE OF CALIFORNIA DO ENACT AS FOLLOWS:

SECTION 1. Section 106.3 is added to the Water Code, to read:

106.3. (a) It is hereby declared to be the established policy of the state that every human being has the right to safe, clean, affordable, and accessible water adequate for human consumption, cooking, and sanitary purposes.

(b) All relevant state agencies, including the department, the state board, and the State Department of Public Health, shall consider this state policy when revising, adopting, or establishing policies, regulations, and grant criteria when those policies, regulations, and criteria are pertinent to the uses of water described in this section.

(c) This section does not expand any obligation of the state to provide water or to require the expenditure of additional resources to develop water infrastructure beyond the obligations that may exist pursuant to subdivision (b).

(d) This section shall not apply to water supplies for new development.

(e) The implementation of this section shall not infringe on the rights or responsibilities of any public water system.

Figure 1.2. Assembly Bill No. 685 and California Water Code Section 106.3. Original text of bill and approval of the human right to water in California.

These features of affordability are not always clearly articulated in the existing literature, and we suggest several ways for researchers to clarify and improve measuring water affordability. In Chapter 3, I apply lessons from the review in the process of developing water affordability measures in the context of California’s right to water. In other words, we develop metrics and discuss how these metrics do or do not represent core dimensions of water affordability. This situates our assessment of affordability in water systems across the state in conversation with the normative aims of the right to water and SDGs.

A novel contribution of this work is that the development of indicators in this dissertation is part and parcel of institutionalizing and developing the context for deeper realization of the right to water in practice. Alongside OEHHA, several concurrent efforts to implement human right to water aims are underway across the state, including a Low Income Rate Assistance program (State Water Resources Control Board 2019b). In 2019, the state approved 1.3 billion USD over the course of ten years to implement programs increasing the safety and affordability of drinking water statewide (State Water Resources Control Board 2019a).

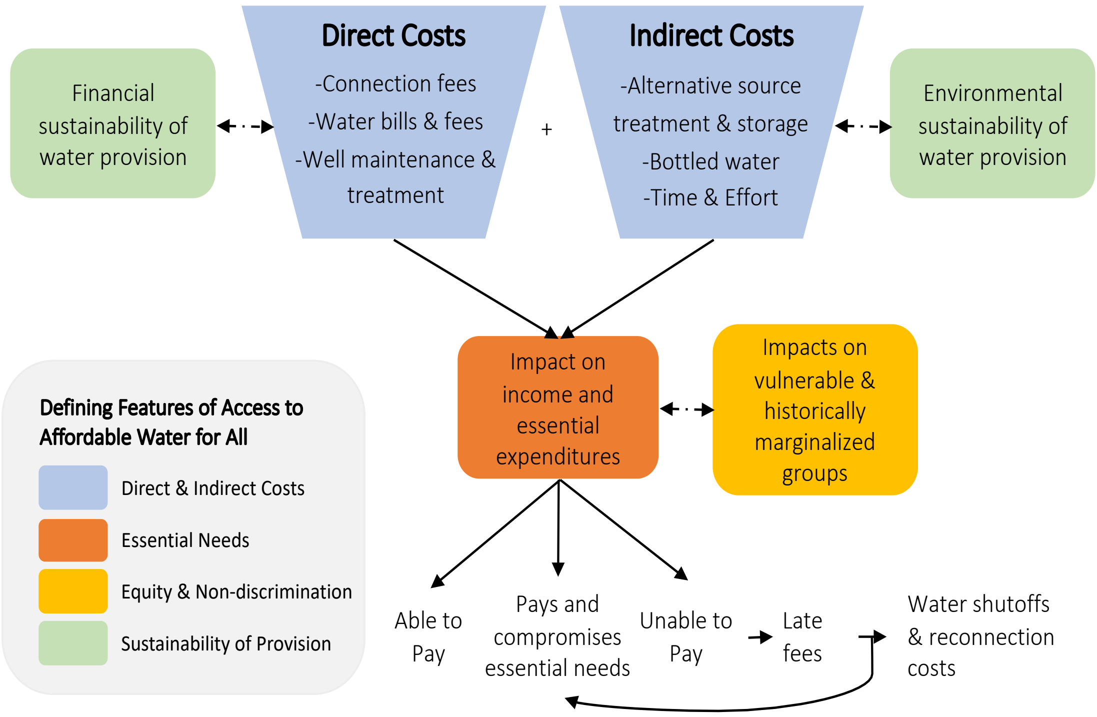

Figure 1.3 Four features of water affordability centered on aspects that impact household affordability.

Distilled from the human right to water, SDGs, and utilities contexts in the U.S. these are: 1: Indirect and direct costs of safe water are affordable to households; 2: Cost of safe water to households does not compromise essential needs; 3. Equity, non-discrimination, and attention to historically marginalized groups; 4. Sustainability (environmental and financial) of water provision.

While these chapters isolate affordability for analysis, the metrics and concepts presented are inextricably linked to a holistic view of water quality and accessibility in the human right to water work in California. Distilling the key aspects of metrics in an academic exercise is essential for developing carefully thought out metrics and assumptions. In practice, however, the lessons and tools of this dissertation are integrated with a broader holistic view of water access.

1.6 SUSTAINABLE WATER ACCESS

The overarching SDG framework clearly embeds the right to water in sustainability aspirations, but like the human right to water, *how* and in what ways these aspirations manifest is rather open. This is in part because sustainability–like water affordability–is normative, and its implementation into policies, indicators, and practice are context-dependent. It is clear, however, that sustainability transitions require a dramatic decoupling of consumption and harmful environmental impacts. Calls to reduce greenhouse gas emissions to net zero by the mid-century globally relay the critical role of decarbonization across all SDGs (Sachs *et al* 2019).

Recently, scholars have proposed evaluating interactions between the SDGs and the 169 targets set forward to meet all 17 goals. Interactions in this new literature are understood as synergies and trade-offs among SDGs (Pradhan *et al* 2017). SDG 12–Responsible consumption and production–has been identified as a cross-cutting goal that would lift the global agenda, due in part to the fact that unsustainable consumption and production in a carbon-intensive energy system presents inherent trade-offs with other SDGs (Pradhan *et al* 2017). Achieving this "lift" in the case of water access may happen partially through meeting existing targets (water use efficiency or pollution reduction). However, the targets are limited in that they were not designed to capture synergistic/antagonistic relationships across goals.

1.6.1 *OVERLOOKED: BOTTLED WATER IN THE SDGS*

An example of a potentially antagonistic relationship between water access and sustainability is bottled water use, which Chapter 4 details in the case of Mexico. The role of bottled water in achieving safe water access is not well problematized in the SDG and safe water access literature. The Joint Monitoring Program (JMP) uses a ladder framework to monitor water access, the SDGs, and the progressive realization of the human right to water. The top of the ladder defines "safely managed" drinking water as access to readily available, on-premise water from an improved source. Until 2017, an unimproved source has been defined as an uncovered well, surface water source, cart or tanker, or bottled water (Weststrate *et al* 2019). Today, however, the JMP includes "packaged or delivered water" as an improved source (WHO/UNICEF JMP 2019b). Packaged water–which includes bottled water–is aggressively marketed by bottled water corporations as a solution to the broader water access challenge. Households with and without tap water access are consuming plastic bottled water at an increasing rate in low-middle income countries (Rodwan 2019). Cohen and Ray 2018 recognize bottled water use as essential where it is the only safe alternative, but they forcefully argue against this trend as a viable pathway to safe water access (Cohen and Ray 2018). What are the implications of including bottled water as a safe water access strategy?

There are clear water quality and affordability implications. The use of packaged or bottled water has been a coping strategy in communities without access to water (Nastiti *et al* 2017b), where relying on a patchwork of multiple water sources is a necessity of daily life. There is an assumption that bottled water is safer than unimproved sources or tap water, due in no small part to significant marketing and advertising budgets promoting the health benefits of bottled water. Yet few studies analyze the water quality of bottled water systematically. Water sold in sachets in West Africa were associated with lower incidence of diarrheal disease (Stoler 2017).

More research has focused on people's choice of bottled water as an alternative to tap water. In this work, bottled water choice is consistently associated with the belief that bottled water is safer. This trend appears to persist even where water access is high, like the U.S. (Pierce and Gonzalez 2017a, Javidi and Pierce 2018). The per unit cost of bottled water is typically more expensive than tap water or household treated water. In areas where water access is lacking entirely, however, communities may find bottled water to be more affordable given the low capital cost compared with new infrastructure (Walter *et al* 2017). The actual affordability impacts on households, however, is not well studied.

Taken together, bottled water is often more expensive than tap water, and water quality is not clearly established as better–but perceptions persist that it is. The JMP recognizes bottled water as an improved source only if quality is high. But this is not easily monitored, because corporations are not held to public scrutiny or obligated to human right to water and SDGs. This raises broader questions about the institutions wielding responsibility for ensuring safety and affordability of bottled water. The tools of human rights, environmental justice, and the question of process freedoms allow us to critically question the role of bottled water in the SDG framework.

The increased reliance on bottled water globally will contribute to a view that more people have safely managed water access. Where bottled water is the only source of drinking water, it is hard to dispute its value. However, if we take the "interactions" lens to this trend, we might observe that the increase in bottled water use as a primary drinking water source presents a negative interaction or trade-off between SDG 6 and other core Sustainable Development Goals like SDGs 11 (Sustainable cities and communities) and 12 (Responsible consumption and production). Previous analyses of the life cycle impacts (climate change and environmental effects) of bottled water indicate high energy use and greenhouse gas intensity (Gleick and Cooley 2009, Fantin *et al* 2014) compared with larger water supply infrastructure (Stokes and Horvath 2006). At a minimum this would suggest that research on bottled water consumption and sustainability are critical to understand the nature of water access (Walter *et al* 2017). Without indicators or language to discuss these interactions, the current SDG agenda and monitoring efforts risk encouraging an unsustainable pathway to water access.

1.7 DRINKING WATER ACCESS IN MEXICO

In Chapter 4, I aim to quantify the climate impacts of bottled water use as a household water source. Mexico has a high percentage of households with access to piped water (96% as of 2017), but this water is not reliably safe. Only 42.9% of the population is estimated to have access to safely managed water free from contamination (WHO/UNICEF JMP 2019a). In this context, bottled water (or 20 liter "garrafones") has grown as the primary drinking water source for most households across Mexico (INEGI 2017). Even where piped water is readily available, deep distrust in public water supply persists (Erickson 2012). The belief that bottled water is superior to tap water is captured in national statistics annually. In 2017, 69% of respondents in the Mexican census reportedly choose tap water because it is healthier, and 5% state they choose it because it is their only source of drinking water (INEGI 2017). Few studies quantify the actual impacts of this pathway in the context of realizing SDGs and the right to water.

The case of Mexico offers a unique look the impacts of bottled water use as a water access strategy for two reasons. First, Mexico has the largest per capita consumption of bottled water in the world, and the use patterns in Mexico are likely to provide insight into the impacts of bottled water use in other emerging economies. Mexico has high penetration of very small or "micro" sized water bottle purification plants (estimated at 52% of the market share). At the same time, reuse of 20 liter bottles known as garrafones is high. These trends suggest areas where emissions may be higher or lower than existing studies on single use-plastics or in high income countries like the U.S. They also reflect drinking water patterns of high reuse and short transportation distance, which remain understudied but likely common in other emerging economies (Dzodzomenyo *et al* 2017). Secondly, Mexico has a national level climate change abatement strategy that has diverted funding and resources towards quantifying sustainability impacts of drinking water.

Mexico's Special Climate Change Program (PECC) set forward an ambitious national commitment to reduce greenhouse gas (GHG) emissions to 50% of 2000 levels, or 340 Mt CO_2eq by 2050 (Commission on Climate Change 2009). The country's National Population Council estimates that housing infrastructure will more than double between 2005 and 2030, and more than seven million new housing units will be constructed between now and 2020, contributing up to 33 Mt CO_2e to the country's overall GHG emissions (SEMARNAT n.d.). As Mexico's residential building sector accounts for 26% of its national electricity use (14% of GHG emissions), and its demand is projected to grow at 5% per year, the residential sector is thus a critical target for GHG abatement measures.

One mechanism for reducing residential sector GHG emissions has been the development of green lines in commercial banks to finance energy efficiency projects (IADB 2012a). The Ecocasa Program was the first large-scale pilot under Mexico's climate abatement strategy, submitted by the government of Mexico to the COP16 in December 2012. The general objective of the Program was to contribute to the reduction of GHG emissions in the housing sector in Mexico by financing housing developers to build housing projects that meet GHG reduction goals established by the Program, as well as provide individuals with mortgages that follow the sustainability criteria established by the National Housing Commission. Following a "whole-house approach" for reaching emission reductions, the Mexican government contracted with the Inter-American Development Bank to develop metrics for emissions associated with all aspects of a household, including domestic water use.

In 2013-2014, I worked with co-authors Dr. Fermín Reygadas, Dr. Diego Ponce de Leon Barido, and John Pujol to develop a set of emissions factors characterizing the full residential water cycle through a project aimed to support the Ecocasa Program, funded by the Inter-American Development Bank. In Chapter 4, I present research findings from our assessment of bottled water use as a primary drinking water source in Mexico.

Decoupling of consumption from energy use are important principles relating to SDG 12 (Sachs *et al* 2019). The energy costs of water use at home and via distribution systems is tightly linked to energy consumption in small and large-scale water systems. Typical energy costs include pumping water through treatment facilities and electricity use for water treatment processes like reverse osmosis and ultraviolet treatment (Cooley and Wilkinson 2012). Water losses (system losses, inefficient household appliances) are considerable energy expenditures born by water utilities, municipalities, and governments (Goldstein and Smith 2004). Although residential water supply withdrawals are small compared to those in the industrial and agricultural sectors, they can have higher energy requirements because water must be treated to acceptable water drinking standards and pumped over longer distances (Robertson 2011, Sanders and Webber 2012).

Life cycle assessment (LCA) is a key tool to analyzing the emissions associated with water use. Environmental engineers and economists developed LCA as both a tool and a perspective to capture economic, environmental, and social dimensions of technology and product design (Yves *et al* 2004). LCAs can help identify the most energy intensive component of water system distribution systems (Skipworth *et al* 2002, Lundin and Morrison 2002) or help planners compare costs and emissions associated with different projects (Tsagarakis *et al* 2003) or management strategies (Savic and Walters 1997, Racoviceanu *et al* 2007). Finally, life cycle assessment can quantify embedded and direct energy expenditures incurred in the fabrication, use, and end-of-life stages of a water distribution system (Friedrich 2002, Yves *et al* 2004). The concept of energy expenditures enables representation of direct energy consumption (e.g. electricity or gasoline) and the energy embedded in materials used throughout a systems' life cycle (Skipworth *et al* 2002).

While some studies have evaluated the carbon dioxide equivalent (CO_2eq) emissions embedded in household water use in other countries (Hackett and Gray 2009, Shimizu and Dejima 2012, Kenjay *et al* 2008), little has been done to evaluate such metrics in Mexico (See Centro Mario Molina 2011). To our knowledge, no study exists analyzing the life cycle impacts of garrafones in Mexico. Bottled water energy costs have the added elements of plastics production, distribution between bottling plants and homes, and reuse and refilling processes. Previous studies indicate that bottled water similar to garrafones (5 gallon home and office delivery bottles) have lower emissions than single use plastic 1 liter bottles, but that emissions are nonetheless substantial (Franklin Associates 2009a).

In Chapter 4, I develop a life cycle assessment to evaluate bottled water emissions in Mexico. By attending to the embedded emissions of bottled water use, I identify those processes that have the most potential for emissions abatement. But more importantly, I demonstrate the substantial emissions savings that could be made possible by pursuing more sustainable alternatives (such as tap water or household treated tap water).

1.8 Conclusions

In this chapter I have elaborated key concepts relevant to the right to water and SDGs. Critiques about water as a right largely center on the tractability of the human right in an increasingly financialized water sector, and process questions about who and how rights obligations will be met. Indeed, how water access should be measured generally, and in what ways it intersects with sustainability goals remains an open question. In the chapters that follow, I offer two ways of engaging with indicator development through a focus on water affordability and sustainability.

The contexts I work in are largely part of the institutionalization of the right to water and the SDGs. I aim to create metrics that are useful to actors in state-government capacities in California and Mexico. Co-production is both a set of theories and descriptions of the way that science and policy construct knowledge (Jasanoff 2006) as well as a methodology for producing useable science (Lemos *et al* 2018). At the core of both the theory and practice of co-production is the recognition that research is a social process. As such, research questions and goals reflect particular political engagements. As affordability and sustainability are both normative terms, the definitions and metrics presented in Chapter 3 and Chapter 4 are one instantiation of these concepts. By situating this work in the context of the right to water and the SDGs, globally, I hope to connect the case of water affordability in California and bottled water use in Mexico to the broader conversation on sustainable water access (Chapter 5).

CHAPTER 2

2. A CRITICAL REVIEW OF WATER AFFORDABILITY MEASURES FOR SUSTAINABLE DEVELOPMENT GOALS AND THE HUMAN RIGHT TO WATER IN THE UNITED STATES[1]

Abstract

Human right to water and sustainable development goals emphasize that human well-being depends not just on the quality and physical accessibility of drinking water, but also its economic accessibility. Despite this recognition, governments and academics alike have been hard-pressed to define and measure water affordability. How can we measure affordability, and in what ways do the methods that exist incorporate the normative positions set forth in the human right to water and Sustainable Development Goals? These questions motivate the critical review presented here We first categorize water affordability definitions and approaches to its measurement across academic and gray literatures, incorporating both international and U.S.-based literature. We then identify several dominant definitions (e.g. water affordability as the ability to pay) and corresponding measures (e.g. affordability ratios), as well as some infrequently used approaches (e.g. utility-induced poverty). Our review distills critical themes on how and what to measure for water affordability, while using human right to water and sustainability norms of social equity and ecological sustainability as cross-cutting themes to evaluate these measures. This results in recommendations to improve water affordability measurements while recognizing the trade-offs between ideal measures and practical implementation. Our review emphasizes the diverse ways that affordability measures can incorporate right to water and SDG aims and thereby foster better representation of the water equity challenges requiring action.

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1 This chapter will be submitted to a journal in edited form and has been approved for use in this dissertation by my co-authors, Carolina Balazs and Isha Ray.

2.1 INTRODUCTION

Affordability is a core pillar of the human right to water (UN 2009) and of Sustainable Development Goal (SDG) 6 on universal access to safe and sustainably managed water and sanitation (UNDP 2018). In the US, water and sanitation access is neither equitable nor universal (Jepson and Vandewalle 2016, Wescoat *et al* 2007, Deitz and Meehan 2019, Allaire *et al* 2018). Affordability challenges for drinking water have been documented in California (Christian-Smith *et al* 2013, Pierce and McCann 2015, Hanak *et al* 2014, U.S. Conference of Mayors 2014), Michigan (Rockowitz *et al* 2018), and the US-Mexico border in Texas (Jepson 2014). Joint analyses of water and sanitation affordability across the U.S. indicate a trend towards high unaffordability, especially for low-income households (Mack and Wrase 2017, Teodoro 2018). Rising water costs and persistent poverty levels contribute to affordability concerns. Rubin 2018 estimated that between 1990 and 2015, average water and wastewater prices tripled in the US (Rubin 2018). Increasing water costs are due in part to the compounding influences of old infrastructure, deteriorating water quality, and inefficient water infrastructure sprawl (Pierce *et al* 2019). These costs affect communities differently, with smaller water systems facing technical, managerial, and operational capacity challenges to meet rising costs (McFarlane and Harris 2018). Higher costs to provision water have contributed to steep increases in water bills for households connected to water systems, raising concern that water affordability is a 'burgeoning crisis' in the U.S. (Mack and Wrase 2017).

Affordability eludes concrete definitions, but the rise in the number of people struggling to pay for their water bill has stimulated interest in clearly defining and measuring water affordability. Scholars from geography, public policy, and microeconomics have measured the extent to which water costs are an undue burden on household incomes (Jepson 2014, García-Valiñas *et al* 2010b, Smets 2009, Hutton 2012, Teodoro 2018, Mack and Wrase 2017). The most common approach to measuring water affordability is the use of ratios that measure the cost of drinking water relative to income, which, compared against a specified benchmark, defines whether water is affordable. The ratio method has several limitations, and researchers and decision-makers are seeking improved measures for affordability in several regions of the world. Such discussions are relevant to meeting the Human Right to Water and SDG 6; both provide clear, aspirational goals on water access, but neither agenda provides consistent guidance on the measurement of water affordability.

The policy relevance of household-level affordability research in the US is significant and growing in response to deteriorating infrastructure and fragmented water management. Between 2017 and 2019 alone, the US Environmental Protection Agency (EPA) commissioned a revisioning of their federal-level water affordability standards (NAPA 2017), the cities of Baltimore and Philadelphia advocated and passed income-based billing for low-income households (Walton 2019), and California proposed a low-income rate assistance program (State Water Resources Control Board 2019b) as well as a tracking and monitoring tool for the human right to water, which included water affordability (Balazs *et al* 2019). These new efforts reflect a revisioning of historical affordability work focused on assessing water system compliance with water quality standards. New dialogue and measures are needed to understand water affordability in a human right to water, and more broadly, SDG paradigm.

In light of this gap between policy goals and existing measures, we conducted a comprehensive review of the academic and gray literatures with the goal of critically assessing how water affordability is conceptualized through its measurement. To do so, we evaluate the potential of existing measures to capture normative aims of the right to water and SDGs for social equity, non-discrimination, and ecological sustainability. Our review connects these broader goals to metrics of affordability identified in the literature and the debates that surround measuring affordability. We focus primarily on the literatures from high-income countries and the US, but identify relevant concepts and tools from the international literature as well. In addition to filling the aforementioned gap, this work ultimately seeks to support researchers and practitioners who develop and use affordability measures

We organize the review by asking: What are guiding frameworks for water affordability? (Section 2.2); How is affordability defined and measured and what are the strengths and limitations of these measures? (Section 2.3); and How do these measures and emergent themes in the affordability literature relate to the guiding frameworks? (Section 2.4). We identify trade-offs between theoretical ideals and practical implementation of measures, and their relevance for emergent themes relating to study scope and scale, and aspirations for equity and sustainability. Finally, we outline future research needs and recommendations for measuring water affordability in a U.S. context, and for policy efforts to mitigate affordability challenges (Section 2.5).

2.1 Inclusion Criteria for Review and Assessment Criteria

To ensure the comprehensiveness of our assessment, we took a structured approach to reviewing the literature. First, we conducted a Web of Science search for all peer-reviewed papers on "water affordability" in early 2019–we applied no date restrictions to the publication search. In sum, 359 articles (excluding conference proceedings and books) were downloaded from Web of Science. Of these, we reviewed titles and abstracts and marked them "Likely Include" (n=81), "Potentially Include" (n=5), "Exclude" (n=275), or "Duplicate" (n=4). Of the 86 articles that were "Likely" or "Potentially" inclusions, 24 were excluded after full text review and 1 was excluded because it was not accessible. This resulted in 61 articles relevant for inclusion in the bibliography after a full-text review.

Studies were review if they explicitly discussed water affordability and potential approaches to water affordability measurement. More often than not, water affordability measures focus on water used for consumption, hygiene, cooking, and cleaning; many metrics, however, include sewer charges in their representation of water costs. While drinking water and sanitation affordability are often considered jointly in policy, we focus on 'water affordability' and did not include sanitation in our search terms. We included conceptual papers and empirical studies, as the goal of the review was both to understand what has been done and to critically assess what remains under-developed in the field of water affordability measurement.

We excluded studies on: the cost or affordability for specific water treatment technologies, willingness-to-pay for hypothetical water rates or hypothetical water system upgrades, and water costs or prices that do not explicitly discuss water affordability. Several comprehensive reviews covering willingness to pay exist (Amrose *et al* 2015, Whittington 2010, Ahuja *et al* 2010). These themes are important, but do not define, operationalize or measure affordability directly. We also excluded survey-based studies on people's perceptions of whether or not their water was affordable (Patel *et al* 2010, Koehler 2018); this literature rarely includes quantitative measures of affordability (an exception includes the water insecurity index literature (Jepson 2014, Wutich *et al* 2017)). Qualitative assessments of this kind are useful in identifying drivers of affordability issues, but do not speak to the measurement of affordability itself.

Secondly, we manually added papers from our own databases that did not surface in the Web of Science search but were identified as relevant to the inclusion criteria discussed above. Third, we complemented these searches with a gray literature review of United Nations (UN) agencies, the WHO / UNICEF Joint Monitoring Programme (JMP), Environmental Protection Agency (EPA), and other prominent institutes' studies and reports. As we were aiming for a comprehensive review with a critical focus, we included these additional papers and reports for richness and insight from the applied literature. This resulted in an additional 47 academic journal articles or reports from the gray literature.

Combining the Web of Science search (n=61 included) with these additional reports and publications, we reviewed 108 articles for their definitions, measurements, and uses of the term "water affordability". We did not exclude studies based on geographic location, because research on economic access to water in low-income contexts is better established in several disciplines. However, we focus the review on papers and findings relevant to U.S. and middle-high-income country contexts.

A diverse set of research disciplines engages in the assessment and measurement of water affordability. Disciplines with the most water affordability publications are in the international literature on water sanitation and hygiene (known collectively as WASH) or in the utility policy and management realm–but the range of journals spans sociology, human rights, geography, urban studies, and economics (See Appendix A for full list of papers).

2.2 Frameworks and Normative Aims for Water Affordability

Affordability is an intuitive concept, but it is difficult to define. Both the academic and gray literature tend to begin with measures–rather than with conceptual definitions–of affordability. Nonetheless, we identified three institutional contexts especially relevant to the U.S. where water affordability is defined and measured. These are the human right to water, the SDGs, and EPA's affordability framework in the US.

2.2.1 *Identifying Key Frameworks*

The UN General Comment No. 15 (GC15) set the stage for the human right to water worldwide (United Nations Committee on Economic Social and Cultural Rights 2002, UN 2005, Salceda *et al* 2013). Water and sanitation are economically accessible, according to the GC15, if their direct and indirect costs are affordable and do not impact a person's access to other essential rights (e.g. food or shelter). GC15 does not explicitly define affordability. The Comment indicates that economic accessibility, and thus water affordability, overlaps with three other dimensions of water access–

physical accessibility, non-discrimination, and information accessibility. In other words, affordable but unsafe or inaccessible water is not acceptable. Service disconnections consequent to inability to pay are thus a violation of the GC15's concept of water affordability. The goal of non-discrimination in GC15 imparts a focus on vulnerable and historically marginalized groups.

The Sustainable Development Goal 6, and the targets associated with it, is the most widely accepted approach to operationalizing key aspects of the human right to water (Gawel and Bretschneider 2016). The Joint Monitoring Programme for Water Supply and Sanitation (JMP) operationalizes measures of water access largely through a drinking water service levels lens, which quantifies whether water is of adequate quality and accessibility (Grigg 2018, Kayser *et al* 2013, WHO/UNICEF JMP 2017). As of 2019, definitions for physical access and quality populate the water service ladders; while affordability is acknowledged, its definition and incorporation of affordability remain under development and are not yet formally represented. However, the service ladders do incorporate a measure of time costs associated with acquiring water–basic water service is defined as needing to travel no more than 30 minutes to obtain water (WHO/UNICEF JMP 2017). As noted in the introduction, the SDGs extend the human right to water framework by grounding water access in a broader framework for ecological sustainability. Affordability of water is influenced by sustainability goals in several ways, including water rates that charge more to mitigate drought or incentivize conservation (Cooley *et al* 2016) and the water losses that lead to higher water bills due to inefficient appliances and leaking pipes (Bakker 2010). Thus the ecological sustainability of water access can influence affordability at the household level.

Historically, the dominant framework for considering water affordability in the U.S. has been one focused on assessing the financial impacts of water service provision and water quality compliance on households and water systems. EPA's affordability framework emerged after the 1970s to help water systems coming into compliance with Safe Drinking Water laws. The framework provides guidelines to states disbursing Safe Drinking Water Revolving Funds and has two parts: states should 1) measure water affordability as the ratio of water (or wastewater) bills to median household income within water systems, and 2) estimate a variety of financial capacity indicators of the community served by a water system (US EPA 1997, 1998a).

Utilities in the U.S. have long focused on the link between household affordability and water system-level financial capacity (Davis and Teodoro 2014). Financial capacity is partially a function of the stability and predictability of revenue (Blanchard and Eberle 2013), which often derives from consumer rates and fees. Water systems reliant on household revenue (as opposed to taxes or transfers) can be impacted by household-level affordability in two ways: households with affordability challenges may go into arrears (thereby reducing the utility's revenue), and rate increases for consumption or conservation interventions may lead to reduced consumption (thus reducing revenue). Low financial capacity can lead to unsustainable operations that fail to address water quality concerns, thereby increasing the burden of unaffordability for households forced to purchase bottled water (due to poor tap water quality) or to pay high bills following a rate change implemented to cover compliance costs. This pattern has been documented in the US for decades (Jones and Joy 2006, US EPA 1998a).

After several decades of rising water costs compounded by deteriorating infrastructure, the federal framework has been challenged as inadequate to capture low-income households plight (NAPA 2017). This reflects a broader shift toward consideration for household water affordability in the U.S., and emphasizes that new criteria is needed to advance concepts for water affordability in the U.S. The human right to water and Sustainable Development Goals offer a broad perspective connected to larger debates on water access globally. Several states have adopted the human right to water as a guiding framework for state-level policy. California has formulated affordability as a human rights issue and Massachusetts and Pennsylvania have embedded the principle of affordable water into their constitutions (Balazs *et al* 2019, State Water Resources Control Board 2019b, Constitution of the Commonwealth of Massachusetts n.d., Pennsylvania. n.d.). Nonetheless, a utilities-perspective that emphasizes financial sustainability remains the dominant arena in which household affordability gets attention in the U.S.

2.2.2 *Defining Features*

Several defining features of water affordability emerge from these frameworks and goals. Figure 1 illustrates four defining features of what affordability is, and which normative aims from the SDGs and human right to water relate to affordability (Figure 1). While all four features refer to household-level access, these aspects of water affordability are frequently conceptualized from a community or utility perspective, especially equity and non-discrimination and the sustainability of provision.

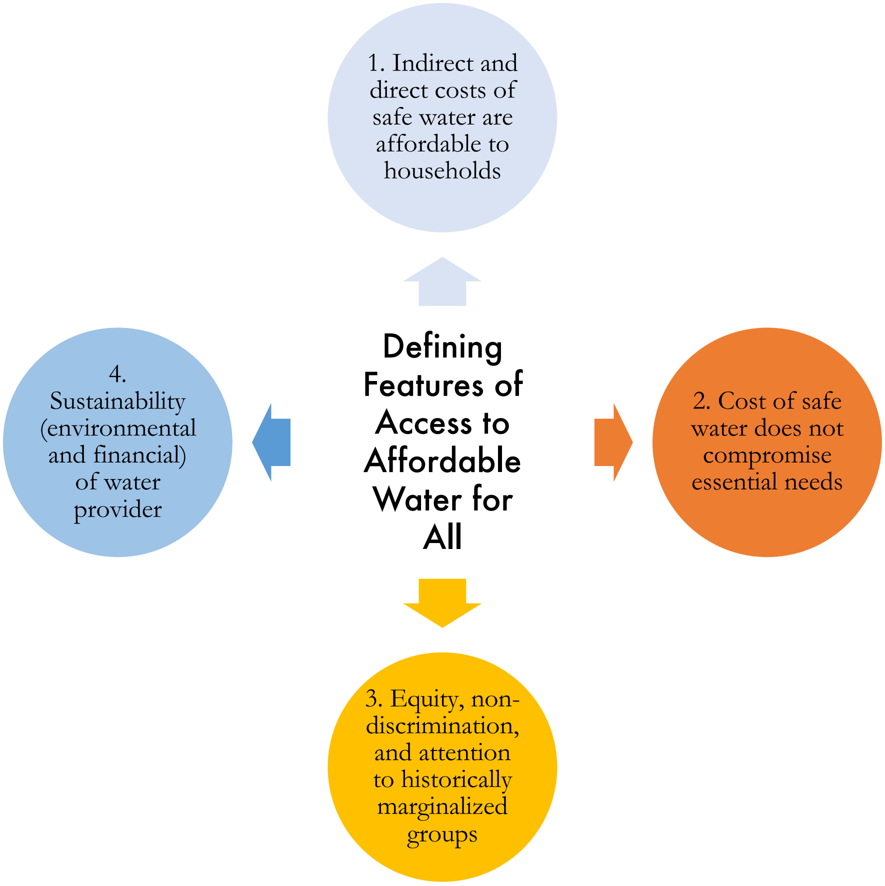

Figure 2.1 Four defining features of water affordability.

A key difference between the human right to water/SDG aims and the EPA/utility focus is the relative emphasis on social versus economic equity. Social equity most often refers to disproportionate burdens of ability-to-pay among different groups based on for example, income levels, socio-economic status, or region. Economic equity, or the classic welfare economic concept that people should pay equally for receiving the same benefit ("benefit principle"), underlies arguments for full cost-recovery and relates to the financial sustainability of provision (Bakker 2010). Affordability poses both social and economic equity challenges, but the context of application (e.g. a utility versus an SDG benchmark or human right to water metric) influences the extent to which both forms of equity are engaged and discussed together, if at all. A comprehensive treatment of various equity concepts is beyond the scope of this review (Davis 2005), but we focus primarily on social equity as it relates to household and community level affordability.

2.3 How is Affordability Defined and Measured?

Defining affordability is not a straightforward task, but it is has to do in part with defining the "reasonableness of burden" of acquiring safe water for households and communities (Gawel and Bretschneider 2016). Measuring this burden entails identifying how much it costs to access water (whether through piped water or via other sources) and what resources households have available. Criteria to determine if the measured burden is too much, or too little (i.e. affordable, or not) varies widely across the literature. The following sections organize findings from the review on: 1) measuring water costs, 2) common definitions of and approaches to measuring affordability (i.e. affordability ratios), and finally, 3) more recently proposed metrics.

2.3.1 *MEASURING WATER COSTS*

Before covering measures of affordability, we review a key component of affordability measures–water costs. GC 15 articulates that both direct and indirect costs of water should be affordable (**Fig 2.1 →*Direct and Indirect Costs***). Direct costs are the costs of water service. Water systems determine household level water bills typically through rate structures, which depend on the existence of a meter (to measure water consumption) and a block or tiered approach to pricing. These rate structures are only part of a water bill, however. Subsidies, life-line rates, and fees (e.g. for wastewater and stormwater) also contribute to bill calculations that comprise direct costs.

Indirect costs are the costs associated with acquiring safe drinking water outside of the primary water service provided. Indirect costs can be financial or non-financial, and they include the time, effort, labor, and financial costs to collect, treat, store and dispose of water from a variety of sources, including purchasing bottled water (Burt *et al* 2018, Narayanan *et al* 2017, Amit and Sasidharan 2019, Soares *et al* 2002). Of course bottled water–and any other non-tap water source–can be considered direct costs if this is the primary water source. Many of these costs are considered coping or mitigation costs because they relate to a lack of access to adequate safe water (Zawahri *et al* 2011, Amit and Sasidharan 2019, Baquero *et al* 2017, Banerjee and Morella 2011). While the distinction between direct and indirect may seem somewhat arbitrary, in context this distinction can help clarify where costs are *additional* to a primary source. We use the language represented in the literature reviewed but acknowledge the limitations of this terminology. We organize the literature findings first by describing common coping and replacement cost categories, followed by a section on their measurement. Then we discuss measures for costs associated with debt (arrears) and water disconnections.

2.3.1.1 DEFINING COPING AND REPLACEMENT COSTS

Households often adopt coping behaviors when they lack access to a reliable water source altogether or when a household's main source (e.g. piped supply or a private well) does not provide safe water. Unaffordable connection costs can result in households (Howard and Bartram 2003, Jimenez-Redal *et al* 2014, Mason 2014, Narayanan *et al* 2017) and communities (Balazs and Ray 2014) being unable to access safe drinking water. Coping costs like time, effort, pumping, storage and treating new water sources can put large burdens on households (Pattanayak *et al* 2005, Pickering and Davis 2012). These costs are better documented where water supply is intermittent (Nastiti *et al* 2017a, Burt *et al* 2018) or where households lack access to piped water altogether (**Fig 2.1→*Equity & Non-discrimination***). Hutton 2012 outlines several factors for consideration in understanding these coping costs, including capital expenditures and maintenance costs of all water, sanitation, and hygiene.

Where water is unsafe, perceived as unsafe, or inaccessible via a piped connection, households will purchase water from other sources or implement household treatment. A large number of potential sources exist, including vendors, kiosks, neighbors with connections, standpipes (i.e. public shared taps), and bottled water. Studies based in low- and middle-income contexts have attended more to the costs associated with different water sources than those in high-income countries like the US. For example, Srestha et al. 2017 include the sum of expenditures for piped water, jar water, and water delivered by tanker for households relying on multiple sources in Nepal. Households may augment expensive piped supply by pumping groundwater (Nastiti *et al* 2017b) or using standpipes (Zuin *et al* 2011), even though the latter option may be more expensive than piped supply where access is limited (Banerjee and Morella 2011). Relatedly, Walter et al. found that study participants in Indonesia perceived expensive bottled water as more affordable than piped water supply because of the combined time and financial burden associated with treating unsafe tap water (Walter *et al* 2017). Such trade-offs indicate the sophisticated cost calculations households make–weighing knowledge about water quality, access, and the time, effort, and money burdens of these options.

Regardless of the primary drinking water source, bottled water use is increasing globally (Cohen and Ray 2018). Frequently, households perceive bottled water as a safer alternative to tap water, even if these claims are based more on mistrust in utilities (Javidi and Pierce 2018) than in the actual safety of bottled water. In some contexts, bottled water is the primary drinking source because it really is the only safe option (Walter *et al* 2017, Moore *et al* 2011, Komarulzaman *et al* 2017). Studies evaluating affordability of bottled water have primarily been based outside of the U.S., though the U.S. is one of the largest consumers of bottled water.

2.3.1.2 Measuring Coping and Replacement Costs

Though our review identified several studies assessing and monetizing coping costs, their inclusion in affordability metrics is uncommon in industrialized contexts. Addressing these costs is somewhat straightforward for financial costs. Capital expenditures like purchasing treatment units and the ongoing costs of maintenance can be collected and amortized over the expected life-time of the investment to estimate the financial burden of storage, treatment, and pumping (Pattanayak *et al* 2005, Amit and Sasidharan 2019). To capture the cost of using alternative sources and related coping costs, some studies have measured the 'replacement costs' (Moore *et al* 2011) or 'combined costs' (Walter *et al* 2017) of bottled water and tap water in households. Moore et al. 2011 include the measured bottled water costs into conventional affordability ratios and found the added costs put an unreasonable burden on households' incomes. The purchase of water from vendors (e.g. grocery store kiosks or small water stations) is usually more expensive and time-consuming than tap water supply, but only two studies in our review evaluated the affordability of vended water (among other coping costs) in the U.S. (Moore *et al* 2011, Christian-Smith *et al* 2013).

Measurement options are less clear for non-monetary costs. Distance for one trip to collect or purchase water, frequency of trips, and the presence of a water hauler are relevant data for capturing time and effort costs around drinking water (Hutton 2012).Time spent collecting water can be estimated as a 'time cost' to households (Zuin *et al* 2011) and then monetized by estimating lost wages associated with the time lost–e.g. by multiplying the time spent accessing water by the hourly wage (Amit and Sasidharan 2019) or 50% of the hourly wage (Pattanayak *et al* 2005), where the prospects of employment are low. The Joint Monitoring Programme service ladders characterize basic access as water that takes less than 30 minute round-trip to acquire and limited access as water that takes over 30 minutes round-trip.

Several costs are not well-captured in the literature reviewed. Household water treatment costs and/or the creation and maintenance of a well are underrepresented in the affordability literature. However one case study in Indonesia estimated the costs of water boiling and well-drilling–what they called 'mitigating' costs for poor or inadequate water supply (Nastiti *et al* 2017b). Costs were amortized to capture the capital and operating costs for these alternative sources. No studies in the U.S. or industrialized regions were identified that captured the costs facing well-owners, whose water quality is not regulated or monitored.

We want to note that literature outside the scope of this review dates back to the 1980s and 1990s quantifying coping (or "aversion") costs in communities responding to water quality concerns in the U.S. However in these studies, reviewed in (Whitehead *et al* 1998), affordability is not measured or discussed–except in a case of a *Giardia* outbreak in Pennsylvania in which the authors note that affordability may be a problem (Laughland *et al* 1993). The aversion cost literature has focused on coping mechanisms as providing insight into communities willingness–rather than ability–to pay.

2.3.2 *Arrears & Disconnection*

Another set of costs relates to the costs associated with maintaining water access. Households struggling with water affordability may be unable to pay their water bill on time or at all (Beecher 1994). Non-payment of water and sewer bills results in arrears, with the risk of service disconnection. Additional fees and re-connection costs can occur for households with high arrears; in Baltimore households or community centers with high arrears have been placed on tax-liens (Colton 2017). Disconnections subsequent to inability-to-pay are a violation of the human right to water, but persist as a practice of utility management nonetheless. While international studies clearly document arrears and indicate service suspensions or temporary disconnections (Fankhauser *et al* 2008), U.S.-based work on disconnections, and the costs incurred from them, remains sparse. However, advocacy efforts to raise awareness about the phenomena after cities like Detroit shutoff water to tens of thousands of homes in 2014 and 2015 (Jones and Moulton 2016) motivate the need for more studies.

Several metrics exist to approximate the burden of arrears and disconnections as components of affordability problems. These include: water shut-offs, the number of late bills, the amount of arrears, and the frequency of customers with recurring payment problems (Fankhauser and Tepic 2007). Proposed measures to capture disconnections include: number of disconnections due to non-payment and a qualitative assessment of protections in place for households (Roaf *et al* 2005). Such metrics have also been proposed for utilities to track the effectiveness of affordability program interventions over time (Hasson 2002).

Quantification of non-payment rates does not always signify unaffordability. For example, Banerjee and Morella (2011) found households in the highest income quintile across Africa had high rates of non-payment (Banerjee and Morella 2011). Yet such a metric used for lower income households is likely to indicate significant affordability risks. Furthermore, non-payment across income levels might be a useful indicator of broader risks to water service providers that rely on payments to operate (**Fig 2.1. →*Sustainability of Provision***). No papers in this review explore the relationship between income and payment delays or defaults in the U.S.

2.3.3 *Ratio Measures*

In the sections that follow, we focus on affordability definitions and measures. The most common definition of water affordability is the ability-to-pay for the cost of water in relation to income (Hancock 1993). In theory, both direct and indirect costs are relevant to affordability (**Fig 2.1. →*Direct & Indirect Costs***). The roots of affordability as ability-to-pay, which treats water as a good among others in a household budget, are in microeconomics and public policy research (Kessides *et al* 2009, Martins *et al* 2016). The scale of analysis for most measures is theoretically at the household–but data constraints mean that the application of measures is more frequently at aggregated levels of analysis (such as a water system).

To measure affordability as ability-to-pay, researchers typically compare the ratio of water costs to income (the affordability ratio) to a pre-defined threshold to indicate whether water is affordable or not. Thresholds to evaluate affordability ratios range from 1.5% - 10% (Table 2.1). The large range of thresholds reflects different assumptions about what counts as a cost (e.g. drinking water only or inclusive of wastewater services) and how to measure income (e.g. disposable or gross income levels).

Table 2.1 Affordability ratio thresholds and applications.

Affordability Ratio Threshold	*Water Cost*	*Income type*	*Scale*	*Refs*
1-2%	Drinking water	Median Household Income (MHI)	Nation; Water System; Census area	(Pierce and McCann 2015, US EPA 1998a, Hanak *et al* 2014, Christian-Smith *et al* 2013)
2%	Wastewater	MHI	Water System	(US EPA 1997)
2.5%	Drinking water	MHI / Gross income*	Nation; Water system; Household	(US EPA 1998a, 1998b, EPA Science Advisory Board 2002)
3%	Drinking water &/or wastewater	Disposable or discretionary income*	Nation; Water system; Household	(United Nations Development Program 2006, Reynaud 2010, García-Valiñas *et al* 2010b, Sawkins and Dickie 2005)
4.5%	Drinking water & wastewater	MHI	Nation; Water system; Census area	(US EPA 1998a, 1997, Mack and Wrase 2017)
5%	Drinking water & wastewater	MHI	Nation; Household	(Villumsen and Jensen 2014, Banerjee and Morella 2011)
10%	Drinking water & wastewater	Discretionary income* (lowest income quintile)	Metropolitan area; Water system	(Teodoro 2018, Feinstein 2018)

*There is not a clear consensus in the literature regarding the use of gross income levels (before taxes or expenditures), disposable income (gross income less taxes), or discretionary income (income less taxes and other essential expenses like housing). Income types reported in the table are illustrative of commonly indicated types in the literature.

2.3.3.1 Conventional Affordability Ratios

The simplest ratio measure is the conventional affordability ratio, which measures direct water costs for average water use in a household or region as a proportion of household income or median income of a region, respectively (Al-Ghuraiz and Enshassi 2005, Hoque and Wichelns 2013, Smith and Green 2005). From a human right to water perspective, all costs of water, inclusive of sanitation and hygiene, should be incorporated into these ratios. In both international and national studies, however, drinking water is often treated separately from sanitation, and indirect costs such as purchasing bottled water (in addition to tap water) are less commonly included in ratios (but see (Nastiti *et al* 2017b, Walter *et al* 2017, Komarulzaman *et al* 2017)). In the U.S., median income levels are incorporated into metrics emphasizing financial sustainability of water systems (**Fig 2.1. →*Sustainability of Provision***), but this is increasingly criticized as being at odds with goals for social equity (**Fig 2.1.**

→***Equity & Non-Discrimination***). Though not a metric, per se, water affordability has been represented by picking an affordability ratio threshold a priori (e.g. 1.5%) and estimating the number of households that have adequate income to purchase water so that it is no greater than this percentage (Al-Ghuraiz and Enshassi 2005, Christian-Smith *et al* 2013, Feinstein 2018).

2.3.3.2 Potential Affordability Ratio

Several advances to the conventional affordability ratio exist (Table 2.2). The potential affordability ratio estimates costs only for a 'basic needs' volume of water, variously referred to as an essential minimum quantity (Martins *et al* 2019) or a lifeline level (García-Valiñas *et al* 2010b). The potential water bill for basic needs is divided by household income or by aggregated income estimates for larger areas (Gawel *et al* 2013, Kessides *et al* 2009, García-Valiñas *et al* 2010b, Miniaci *et al* 2008, Fankhauser and Tepic 2007). The primary advantage of this approach is that it avoids the problem of evaluating water affordability when actual expenditures on water reflect excessive levels of consumption (**Fig 2.1. →*Sustainability of Provision***) or under-consumption (i.e. self-rationing) among households that cannot afford basic needs water (**Fig 2.1. →*Equity & Non-Discrimination***). Applied in multiple areas (where the essential needs water costs may vary), the potential affordability ratio can capture unaffordability driven both by poverty and high water costs (as explored in Chapter 3).

Martins et al. 2019 introduced the ratio gap metric, which measures the difference between a potential affordability ratio and a conventional affordability ratio for a household. A negative ratio gap indicates potential under-consumption (i.e. the expenditure on water is less than essential needs), whereas a positive gap may indicate over-consumption (Martins *et al* 2019).

2.3.3.3 Affordability Ratio for 20th Income Percentile

Davis and Teodoro (2014) introduced a variant of the potential affordability ratio–the AR_{20}–to measure the impact of water and sewer bills on discretionary income for households earning at the 20^{th} percentile income level in the urban U.S. (Davis and Teodoro 2014). Discretionary income is estimated by removing modeled expenditures for food, housing, taxes, medicine, and home energy from the 20^{th} percentile income of metropolitan regions served by large utilities. Studies implementing AR_{20} focus on utilities in the U.S. serving populations larger than 3,300 people (Teodoro 2018, 2019). This approach offers a conceptual advantage over other affordability ratios in that it attempts to indicate whether water costs are unaffordable after other needs have been met (**Fig 2.1. →*Essential Needs***) for low-income households earning near the 20^{th} percentile. The selection of the 20^{th} percentile income requires specification of spatial scale. While the 20^{th} percentile may reflect low-income households in a relative sense (e.g. in areas where cost of living is high and thus the 20^{th} income percentile is a higher), more work is needed to clarify what this looks like in small areas.

As with selecting a poverty threshold in the residual income approach, AR_{20} requires analysts to pre-determine a set of normative criteria–a basic needs amount of water in addition to a minimum budget for other essential goods. Though challenging to implement at scale, this approach addresses several key features of affordability (Table 2.2).

2.3.3.4 Full Economic Costs and Capital Expenditures as a Portion of Income

Hutton 2012 suggests (but does not calculate) a set of ratio measures for nation-scale affordability tracking in a human right to water context. These measures advance the potential and conventional affordability ratios by emphasizing the full financial and non-financial costs of water (**Fig 2.1. →*Direct and Indirect Costs***), as well as one-time costs such as a connection fee. For example, potential and conventional affordability ratios do not address the cost of obtaining a water source connection. Adding connection costs to the numerator of an affordability ratio (full financial expenditure relative to income) can indicate the unaffordability of access (Hutton 2012). This metric captures only the point in time that a household or community incurred the connection cost. Therefore, connection costs need to be amortized over the expected lifetime of the investment in order to estimate a monthly/annual cost commensurable with monthly/annual water bills (Pattanayak *et al* 2005).

Hutton expressly recommends that indicators should be developed and disaggregated by multiple categories wherever possible, including but not limited to wealth and income quintiles, median incomes, low-incomes, employment status of households, ethnic/racial groups and by households living with disabilities (**Fig 2.1. →*Equity & Non-discrimination***) (Hutton 2012).

2.3.4 *Alternative Measures*

In addition to the many variations of affordability ratios discussed above and in Table 2.2, alternative measures of affordability aim to focus on affordability as it relates to poverty (utility induced poverty).

2.3.4.1 Utility Induced Poverty & the Residual Income Approach

Utility-induced poverty defines water and other utilities as unaffordable if households fall below the poverty line after paying for all utilities (Miniaci *et al* 2008). This work extends analyses of water affordability ratios in the context of a household's ability-to-pay for all utilities, including heat, gas, and electric (Fankhauser *et al* 2008, Fankhauser and Tepic 2007, Mohlakoan and Dugard 2017). Gawel et al. 2013 and Miniaci 2008 applied the utility-induced poverty concept with the residual income approach method, which measures water as unaffordable when the cost of water puts a household's residual income below the poverty line (Gawel *et al* 2013, Miniaci *et al* 2008). Like the affordability ratio, the residual income approach was first proposed in the housing affordability literature (Stone 1990, Kutty 2005). Another variant is the "double" residual income approach, where water is also considered unaffordable if a household's income less all essential expenditures (**Fig 2.1. →*Essential Needs***) falls below the hypothetical cost of paying for a basic needs amount of water. Connection costs can also be included in the bundle of essential needs estimation (Kessides *et al* 2009). While similar to AR_{20} in its attention to lower income levels and essential needs, the residual income approaches differ from other metrics reviewed in that households in poverty are deemed as having unaffordable water regardless of their water costs.

2.3.4.2 Hourly wage

Davis and Teodoro (2014) proposed calculating the number of hours worked at minimum wage (HM) to afford water and sewer bills (Davis and Teodoro 2014, Teodoro 2018). The authors suggests that an affordability challenge exists if HM is greater than 8 hours (or a full day of work in the U.S.). Where ratio based measures aim to capture water cost burden on income, hours at minimum wage aims to capture the time it would take someone working at minimum wage to earn the equivalent income necessary to pay their water bill. This approach is notably easy to calculate because minimum wage levels are readily available by state or city. Teodoro (2018, 2019) has implemented this approach to complement AR_{20} as a two-pronged measure affordability.

Reframing the question of ability-to-pay into a question of reasonableness of labor time required to pay does not directly address ability-to-pay and is perhaps better understood as a way of measuring water costs rather than affordability. To illustrate this, consider a household that has a working member fully employed at minimum wage for 40 hours a week every week, and another household with a family member employed for only half that time. If both of their water bills are less than 8 hours at minimum wage, their water might be considered affordable in this measure. Yet we know from these two households that this says nothing about the two households' ability to pay for water. Conflating affordability with time value of labor also risks normalizing (possibly) inadequate minimum wages. While in theory minimum wage is enough to support cost-of-living in an area, most households living on minimum wage do not have adequate income. This problem is not entirely unique to HM but also applies to monetization of other coping costs described earlier (e.g. time to fetch water).

Table 2.2 Water affordability measures and their link to the human right to water and sustainable development goals. An “x” indicates whether a measure has the potential to address or incorporate a key feature of water affordability.

Indicator Type	*Indicator or Measurement*	*Scale(s)*	*Description*	*Indirect/ Direct Costs*	*Essential Needs*	*Equity/ Non-discrimination*	*Sustain-ability of Provision*	*Key Refs.*
Ratio Measures	Conventional Affordability Ratio	Household Community	Water/wastewater expenditures relative to income‡	x			x	(Mack and Wrase 2017, Roaf *et al* 2005, Hutton 2012)
	Potential Affordability Approach	Household Community	Water/wastewater expenditures for a specified quantity of water relative to income‡	x			x	(Miniaci *et al* 2008, Gawel *et al* 2013, Martins *et al* 2019, García-Valiñas *et al* 2010b)
	Ratio Gap	Household Community	Measures the difference between conventional affordability ratio and a potential affordability ratio	x	x	x		(Martins *et al* 2019)
	Affordability Ratio at 20th Percentile	Community	Water and/or wastewater expenditures for a specified quantity of water relative to 20th percentile income less other essential expenditures	x	x	x		(Teodoro 2018)
	Capital expenditure relative to income	Household Community	One-time costs of capital expenses relative to income‡	x		x	x	(Hutton 2012)
	Full financial expenditure relative to income	Household Community	All financial costs (including water, storage, treatment, on-site sanitation) relative to income‡	x		x	x	(Hutton 2012)
	Full financial and economic costs relative to income	Household Community	Full financial costs (as above) in addition to economic costs (e.g. time to collect and treat water, provide for sanitary needs, and economic value of time) relative to income‡ or expenditures	x		x	x	(Hutton 2012)
Utility-induced poverty measure	Residual Income Approach	Household Community	Defines water as unaffordable when residual income‡ after paying for an essential amount of water is less than the poverty line	x	x	x		(Gawel *et al* 2013, Miniaci *et al* 2008)
	Double Residual Income Approach	Household community	Same as residual income approach, but also removes non-utility essential expenditures from income‡	x	x	x	x	(Gawel *et al* 2013, Miniaci *et al* 2008)
Labor burden	Hours Minimum Wage	Community	Number of hours at minimum wage required to afford water bill	x		x		(Teodoro 2018)

‡Income can be household level reported income or an aggregate measure of income (e.g. median, or across income deciles).

2.4 Critical Themes in Measuring Water Affordability

Affordability ratios measured at aggregated scales (e.g. census tract or water system) are the most common approach for measuring water affordability. There is significantly less attention paid to coping costs, capital expenditures, replacement costs, and indirect costs in the US-based literature. The prevalence of simpler measures, such as water bills as a proportion of median income, reflect the paucity of household-level data on water affordability. Ultimately, this reflects a tension between ideal affordability measures that rely on rich household-level data and specification of relevant costs, and the practical implementation of measurement in data-constrained environments.

This section distills themes that we identify as critical to advancing affordability measures. These themes reflect the judgments and trade-offs inherent in applying affordability measures, and include: the spatial and temporal scale and scope of study, the amount of water required to meet basic needs, the costs that should be included, the income available to households, and the criteria for affordability. We leverage the core features of water affordability introduced in Section 1 to identify how analysts might better navigate practical data constraints *and* incorporate normative aims. These cross-cutting themes are social equity/non-discrimination and ecological sustainability.

2.4.1 *The Spatial and Temporal Scale and Scope of Study*

Water affordability is a metric relevant at multiple spatial and temporal scales of analysis. Microeconomic analyses and human right to water aims describe the household scale as ideal for measuring affordability (Martins *et al* 2016, Jepson and Vandewalle 2016, Gawel *et al* 2013, Vanhille *et al* 2018). This presents a formidable challenge in the policy context and for broad-scale assessment efforts, as household-level data is rarely collected. Such efforts tend to rely on aggregated data. For example, affordability has been measured at the nation scale for human right to water monitoring (Smets 2017) or at the water system scale in the U.S. to disburse Safe Drinking Water Revolving funds to assist systems coming into compliance with water quality standards (US EPA 1998a). When measuring affordability at the community- or water system-scales, measures can signal affordability challenges for a household income stratum (e.g. AR_{20}) where micro-level data is absent. (Un)affordability at the median household income level of a water system or area can signal financial sustainability problems for the entire community (as evidenced by EPA's use of the median household income for several decades). Nonetheless, aggregate measures obscure underlying variability of water costs or incomes within a population. Addressing this critique is critical, and has been met in part where studies stratify aggregate data by income groups and household types (Sawkins and Dickie 2005, OECD 2009) within census tracts (Mack and Wrase 2017), within municipalities (García-Valiñas *et al* 2010a, U.S. Conference of Mayors 2014), or water systems (Teodoro 2018).

Less explored in the literature is the role of temporal scale in terms of trend analysis or rate shocks to household income and ability to pay. Affordability is commonly evaluated using cross-sectional data, but some studies have included a temporal dimension, wherein households' ability to pay for water is assessed given predictions about future water rates and incomes (Mack and Wrase 2017, Fankhauser and Tepic

2007). One report analyzed how drinking water rates increased consequent to California's drought (**Fig 2.1. →*Sustainability of Provision***), during which entire water systems lost access or were forced to drill new wells, passing on expensive costs to households (Cooley *et al* 2016). Rate increases over time are driven by many factors, including environmental change and deteriorating infrastructure. Time series data has already shown that water rates are rising steadily, but deeper analysis that incorporates ability-to-pay and equity across households or communities over time were not identified.

2.4.2 *Water to Meet Basic Needs*

Measuring water affordability requires some specification of the volume considered. Should water be affordable for average levels of consumption, or just for essential needs? Conventional affordability ratios, which use average household water use, risk under- or over-estimating unaffordability (Gawel *et al* 2013). The concern about under-estimation arises because households may only consume what they can afford, even if it is not an adequate amount for basic needs (**Fig 2.1. →*Equity & Non-discrimination***); this has been called the problem of "paid but unaffordable" in the U.S. (Colton 2017). Alternatively, measures of average water use are likely to overestimate affordability problems in countries like the U.S., where 'luxury' water use like landscaping is common practice and the frequent focus of conservation efforts (**Fig 2.1. →*Sustainability of Provision***). The phenomena of self-rationing and unsustainable water consumption merit attention to essential needs use in affordability measures.

Where household level studies are possible, the ratio-gap measure that compares conventional and potential affordability ratios is perhaps the most insightful. However, aggregate measures should focus on basic needs water. This approach has the advantage of identifying inequities across regions trying to access the same amount of basic needs water (**Fig 2.1. → *Equity & Non-discrimination***). For example, studies have found that payments for an essential quantity of water are regressive across income levels within municipalities and water systems (García-Valiñas *et al* 2010a, Martins *et al* 2016).

To determine affordability of the volume of water essential for basic needs, it is necessary to decide what constitutes this minimum volume. The literature offers several approaches to this question: 1) determining basic needs water from demand functions, namely the Stone-Geary function (García-Valiñas *et al* 2010a, Sebri 2015); 2) estimating minimum basic water requirements for universal norm-setting (Gleick 1996, Howard and Bartram 2003); and 3) deriving location-specific estimates of water use based on 'reference budgets'–or the minimum amounts of water needed for washing, cooking, hygiene, and consumption (Vanhille *et al* 2018, Feinstein 2018). Comprehensive metrics should also include basic needs for sanitation and sewer services.

Selecting an adequate volume of water for essential needs has been controversial. The scale for measuring affordability of basic needs matters. In both rate design and affordability measures, choosing a volume for household water consumption as opposed to individual consumption can disadvantage large households, which often consume lower volumes of water per person but larger total volumes. Low-income households tend to be larger (García-Valiñas *et al* 2010b, García-Rubio *et al* 2015, Environmental Law Clinic 2017), and thus households paying for water on a volumetric billing or increasing block rate system may be at risk if the lowest block is set at an inadequate basic needs volume. Implementation of free basic needs water in South Africa in the early 2000s illustrates this risk in practice. Households were guaranteed the use of 6 kiloliters per household per month for free, which in theory was aimed to support providing 25 liters per person per day. This efforts disproportionately underserved larger households, who had to pay for the full water bill after exceeding the minimum volume (Smith and Green 2005). This resulted in inadequate access for basic needs in the most vulnerable households. Research in Nepal and Belgium also found that basic needs water varies by region, suggesting a geographic component to consumption that a single volume level may overlook (Vanhille *et al* 2018, Ojha *et al* 2018). Moreover, losses in the distribution infrastructure and/or appliance inefficiencies may require some households to purchase more water to access an essential needs amount (Environmental Law Clinic 2017).

2.4.3 *Counting All the Costs: Direct, Indirect, and Coping Costs*

The studies in this review largely rely on water bills as a measure of direct costs for water. Several studies incorporate water and sewer costs, which, is aligned with human right aims. Because most households in the U.S. receive a water bill, the relevance of fees and subsidies is high but underexplored in the literature. Measures evaluating subsidy and lifeline rate programs have been proposed in the international human right to water frameworks but were not identified in the literature (Roaf *et al* 2005).

Costs associated with alterative water sources (relative to a household's primary source), coping with poor water quality, or arrears and disconnections, directly implicate the human right to water and Sustainable Development Goals. However, we found little agreement on how or if all indirect and coping costs defined in Section 2.3 can or should be monetized and included in affordability measures. Few studies actually included costs beyond the direct water source into affordability measures. A concrete definition of what counts as affordable for such costs is rarely provided. Yet their contribution to understanding water affordability is clearly significant (Nastiti *et al* 2017b). The international literature provides a rich discussion on coping costs in contexts where households cope with lack of access with numerous coping strategies.

In the U.S., coping costs for purchasing bottled water have been quantified and included in affordability ratios in California (Moore *et al* 2011, Christian-Smith *et al* 2013), but costs associated with disconnections, arrears, or non-financial costs for household water treatment or well-maintenance remain unanalyzed. While Hutton 2012 suggests incorporating these into a single affordability ratio (Hutton 2012), it is possible that showing these costs as complementary or supplemental measures is useful as well. In the U.S., water costs for households relying on domestic wells (water systems serving less than 15 connections and private wells) have not been analyzed. For these households, several costs that might typically be thought of as coping costs are incurred as direct costs to households: time, effort, and the direct costs of electricity for pumping and treatment. More research is needed to identify the full scope of potential direct costs for households in the U.S.

Regarding which costs to count, Nganyanyuka et al. 2014 identified 28 coping strategies for household water access in their case study in Dar es Salaam, Tanzania (2014). It is unlikely that all of these have clear monetary values or to what extent they are generalizable across places, even within Tanzania. Pattanayak et al. 2005 identified five coping strategies in their case study in Kathmandu, Nepal–collection, pumping, storage, treatment, and purchase from non-utility sources–and found that time costs made up 45% of coping costs across their sample (2005). These findings indicate that some coping costs may be more relevant than others, but there is not clear guidance on how to identify the most important costs.

Even excluding the challenge of identifying relevant costs to include in affordability measures for a given context, the exercise of monetizing non-financial indirect costs is controversial. Capturing time or effort costs are borne primarily by women in many low-income countries, which implicates gender equity in the choice to include or exclude time labor in water access (Kremer *et al* 2011). Even among lower-income households, those with higher wages are by definition going to "lose more" for the same amount of time spent (Burt *et al* 2018), because these measures typically measure time in terms of a household's wage, so households with higher wages have more to lose, in a financial sense. The literature also has no consensus on whether to monetize time at all, and, if monetization is desirable, whether to value time at the minimum wage or a fraction of the wage, and what the fraction should be (Ahuja *et al* 2010, Pattanayak *et al* 2005).

2.4.4 *Available Income*

Measuring the ability to pay requires some measure of the resources available to households. A core dimension of affordability is that water is not affordable at the expense of other essential needs. The denominator of an affordability ratio or the income variable in the residual income approach should represent the amount of money a household has available to spend on water. A small but important set of studies in the U.S. demonstrate that many households are paying for water that they actually cannot afford. Families may sacrifice other essential needs–like rent, transportation, and healthcare–before they forego paying for water (Colton 2017, Rockowitz *et al* 2018). These precarious trade-offs between high-cost water and other essential needs are well-known in international literature and likely to exacerbate any existing distress in households (Mason 2014).

An ideal analysis would therefore include what a household has available to spend and what they actually spend on water and other essential expenditures over time. In practice, studies use proxy (and simpler) measures like gross income or expenditures to capture available income. Gross income is likely to over-estimate a household's available income because not all income is disposable (income used to pay taxes or to repay student loans). On the other hand, households may be low-income but have high wealth (e.g. retired communities). Using household expenditures to approximate available income can under-estimate available income because expenditures exclude any unspent income. However, for low-income households, expenditures may be a more accurate measure of available income than reported income due to the variable or seasonal nature of lower-income jobs (Deaton 1997), and any remittances the household may receive.

The limitations of using gross income in affordability ratios have been addressed by evaluating water affordability as a proportion of disposable income (Gawel *et al* 2013, Smets 2009), or disposable income less modeled estimates of essential expenditures, which results in an estimate of discretionary income (Teodoro 2018). In the latter case (AR_{20}), affordability is conceptualized as the impact of water bills on a household's spending after they have paid for other 'essential needs'–shelter, food, taxes, etc. This approach has gained traction U.S. policy discussions, e.g. see (Feinstein 2018, NAPA 2017). Recent applications of the method resulted in negative affordability ratios when modeled expenditures for essential needs were removed from income (Jensen *et al* 2019). This is challenging to interpret and perhaps suggests that where expenditure and income data are both available, the residual income approach might provide a better measure.

2.4.5 *Criteria for Affordability*

What counts as affordable is dependent on the socioeconomic context, as well as on social and political values (Page 2005, Teodoro 2018). In practice, what *gets counted* as affordable water usually depends on the choice of an affordability threshold. Internationally, these thresholds emerged from an analysis of average consumer expenditure on other household goods at the median income level, or in lower-income groups, depending on the focus of analysis (Smets 2009). In the U.S., EPA's 2.5% threshold for drinking water affordability derives from an assessment of what median-level households pay for other basic expenses (based on Consumer Expenditure Surveys), the average replacement costs (such as consuming bottled water), and a motivation to minimize permitted exemptions to the Clean Water Act (US EPA 1998b). The US EPA's stance on representing water affordability as up to 2.5% of the median household income has garnered criticism that this benchmark is too high for lower-income households (U.S. Conference of Mayors 2014). Nonetheless this has been a standard threshold used in recent studies in the US (Mack and Wrase 2017) and Canada (Jenzen et al. 2016). In state policy, however, lower affordability thresholds are common. For example, in California, the State Water Resources Control Board has used a threshold for water bills as 1.5% of median household income to provide financial aid to lower-income water systems (State Water Resources Control Board 2018).

How criteria for affordability thresholds came into existence in the first place is less scrutinized (Rubin 2011), and while threshold-based measures have been questioned altogether (NAPA 2017), few alternative criteria currently exist. Even in the case of the residual income approach, the threshold problem gets transferred to the definition of poverty. At what poverty threshold should utility-induced-poverty indicate an affordability challenge? Some researchers (Sawkins and Dickie 2005, Balazs *et al* 2019) present the distribution of results and interpret affordability as a spectrum from more to less affordable, without normatively declaring water to be 'affordable' or 'unaffordable'. This may be appropriate for tracking and monitoring progress of water as a human right over time.

2.4.6 *Cross cutting themes: Equity and non-discrimination*

Affordability in the human rights context requires special focus on the most vulnerable people and households (United Nations Committee on Economic Social and Cultural Rights 2002). This is most frequently interpreted in the literature as economic vulnerability, reflecting the prevalence of income burden measures. In studies using aggregate data, an area may have affordable rates for households at the median income level but actually have unaffordable rates for low-income households (Fankhauser and Tepic 2007). Underrepresentation of low-income groups has led government expert panels and non-profit organizations to critique the lack of adequate indicators in the U.S, and of the use of affordability ratios for median income levels (Environmental Law Clinic 2017, U.S. EPA Environmental Financial Advisory Board 2014).

In the academic literature, low-income groups have been a focal point for affordability ratio and residual income measures. Most commonly, affordability ratios are estimated for households earning the first or second income quintiles (Reynaud 2006, Gawel *et al* 2013, Teodoro 2018) or across all income levels available (Martins *et al* 2016, Vanhille *et al* 2018, Sawkins and Dickie 2005). By definition, residual income measures identify water affordability as a problem if a household is in poverty or if paying for utilities puts households into poverty.

Non-discrimination is a requirement of providing any human right, including the right to water. Sebri et al. 2015 used concentration curves to evaluate income inequality and disparities in the share of water costs borne by different income levels (Sebri 2015). This approach is both visually compelling and insightful in that it retains the distribution of data and indicates the magnitude of inequality in access. Studies that disaggregate affordability outcomes by categories of vulnerability or discrimination beyond income quintiles are examples of evaluating potential discrimination in access to water. Mack & Wrase 2017 follow calls from Beecher 1994 to incorporate socio-economic factors; they analyze "at risk" census tracts (i.e. those with less than the necessary median household income to afford water and sewer at 4.5% of their income) by social and economic variables such as the percent of households receiving public income assistance, on food stamps, or without health insurance (Mack and Wrase 2017, Beecher 1994). .

Other "high risk" categories important to ensure non-discrimination in water access and affordability include housing type (size, unit type) (Pierce and Gonzalez 2017a, Martins *et al* 2019), ownership of assets (e.g. household owner or renter), renters who do not pay direct bills (Environmental Law Clinic 2017), welfare status (Mangold *et al* 2014), gender, or households served by smaller water systems (McFarlane and Harris 2018). No quantitative studies of affordability measures included gender, despite qualitative research indicating its importance (Mohlakoan and Dugard 2017, Carolini 2012). Affordability ratios are higher (i.e. more unaffordable) among smaller systems in California (See Chapter 3). These studies–and others that explicitly engage geographic distribution or identify geographic hot spots (Mack and Wrase 2017, Deitz and Meehan 2019)–begin to indicate patterns of spatial inequity through their analysis across space or categories of place (e.g. urban versus rural).

Inequities along lines of race/ethnicity are persistent in access to environmental benefits and basic needs in the U.S. and reflect an important research direction to investigate non-discrimination in affordability assessments. Recent work on a representative sample of U.S. water systems found that a weak correlation wherein higher percentages of Hispanic households were correlated with more affordable water (Teodoro 2019). These findings are contrary to broader trends in race/ethnicity around water access, where communities with higher percentage of Hispanics or

Blacks are correlated with more water quality violations (Switzer and Teodoro 2018, McDonald and Jones 2018). To some extent the lack of relationship between race and affordability in Teodoro's study may reflect the limitations of affordability measures if they only capture direct water costs. Allaire et al. 2019 found that Tier 1 health risk violations (i.e. requiring immediate action by utilities) are associated with high increases in bottled water sales in nonrural, low-income communities (Allaire *et al* 2019). Earlier work identified communities with higher percentage of Hispanics are more likely to mistrust their tap water, and both Hispanic and Black households disproportionately drink bottled water over the tap water compared to White households (Javidi and Pierce 2018). Bottled water is more expensive than tap water and thus the nature of affordability and race/ethnicity when it comes to the full cost of water access is still to be determined.

2.4.7 *Cross Cutting Themes: Sustainable Access*

The SDGs clearly embed the aspiration for safe and affordable water within a broader framework of sustainability (Kohlitz *et al* 2016), but as Gawel and Bretschneider (2016) emphasize, the criterion of sustainability is rarely connected to normative aims for the human right to water (Gawel and Bretschneider 2016). Yet ecological sustainability is relevant to household affordability in numerous ways.

Most directly, affordability can be impacted by water rates structured to meet conservation goals or through the selection of basic needs water, as discussed above. Households with less efficient appliances or leaking infrastructure use more water and thus incur higher bills (Environmental Law Clinic 2017, Bakker 2010, Smith and Green 2005). Higher rates to incentivize conservation may unduly burden large households with higher basic needs water use or those with inefficient appliances. Studies that try to evaluate affordability by assessing the cost of acquiring basic needs water and comparing this to the first tier of rate structures (Martins *et al* 2013) are in part analyzing the relationship between sustainability and equity concerns. A thorough interpretation of findings from literature on increasing block rates for conservation is beyond the scope of this review, but an assessment of household costs to consume water in the lowest block can indicate equity concerns as discussed above (Ojha *et al* 2018, Teodoro 2005). Hoque and Wichelns 2013 suggest that the lowest consumption level in a rate structure should be affordable and adequate, but higher consumption levels should prioritize cost recovery goals (2013). Disaggregating ability-to-pay as it relates to the equity structure of rate design or cost recovery might allow for representation of affordability issues alongside sustainability goals in areas with metered supply (Gawel and Bretschneider 2017).

Critical system upgrades necessary to replace aging infrastructure or to respond to environmental hazards (e.g. fire, drought) can also result in higher costs for households (Cooley *et al* 2016). Studies that compare water affordability over time might elucidate the role of environmental change in affecting household affordability. This review did not analyze studies that investigated the affordability of adopting new technologies in water systems, but this literature focuses on comparing systems before and after upgrades and thus could guide future studies looking to investigate this for environmental, rather than regulatory, change (Jones and Joy 2006).

Areas with poor water quality or high distrust in utilities have higher bottled water use (Javidi and Pierce 2018). Bottled water consumption can be a financial burden for households (Christian-Smith *et al* 2013, Moore *et al* 2011) and is ecologically unsustainable as a primary source for drinking (as investigated in Chapter 4). Studies investigating replacement costs are important, but we also need better measures on how much bottled water use is serving as a direct cost for households. We know for example that many communities rely on bottled water long after a water quality emergency is identified (Moore *et al* 2011) and that there are possibly equity concerns along lines of race/ethnicity with respect to who and where bottled water is consumed, as noted above.

2.5 Recommendations & Conclusion

Balancing tractability of measures and theoretical ideals is a challenge in water affordability measurement, as evidenced by the persistent lack of clear consensus and guidance on what counts as affordable in the target setting of SDG 6 or human right to water aims. Our review of the literature aimed to: 1) identify leading frameworks and norms for water affordability; 2) identify and categorize definitions and measures of water affordability; and 3) emergent themes in the application of affordability measures as they relate to core features of water. Here, we distill key take-aways and corresponding recommendations for metric development.

Clarifying measures to serve sustainable development and human right to water aspirations.

While tractable and useful for policy-making, the commonly used affordability ratios have short-comings that are often exacerbated by limited data availability. At a minimum, income based measures need to better theorize and clarify their use of different spatial and temporal scales of analysis, their selection or use of basic needs water volumes, their representation of available income, their attention to issues of equity and non-discrimination, and the criteria used to determine affordability. Our review identified many unique strategies to accomplishing these ends, but rarely employed in the same study.

Researchers have largely sought to improve conventional affordability ratios by: 1) adjusting the numerator to capture the cost of basic needs water–called the potential affordability approach; 2) focusing explicitly on low-income levels within a community; 3) removing essential costs of living (e.g. taxes, housing, food, medicine, healthcare and energy) from the denominator to represent households' available income (AR_{20}); 4) including all financial expenditures related to water for households; 5) including all financial and non-monetary economic costs to households; and 6) measuring affordability ratios at multiple scales to capture a diverse range of households.

Household-level studies can accomplish more depth and nuance of analysis and should thus strive to understand the full costs of water and survey other essential needs costs to inform our understanding of affordability. Aggregate measures can cover a larger area and can shed light on the financial sustainability of an area when, for example, water costs are unaffordable for the median income level. Aggregate studies should disaggregate affordability results by categories relevant to equity concerns (e.g. race/ethnicity) and evaluate multiple income-levels.

Interpreting the equity of water costs and income burdens across spatial scales could be enriched by analyses that indicate how communities have experienced affordability issues over time. Future research should develop measures that identify and represent temporal dimensions of affordability. The focus here could be to provide sensitivity analyses to cross-sectional studies, or to provide insight into how affordability might change in response to rate shocks consequent to system upgrades and infrastructure failure, or climactic events like drought. This type of analysis could potentially illuminate future equity concerns while simulating rate-shocks due to financial or ecological sustainability challenges. Otherwise, real costs (e.g. drought or fire surcharges) could be added into affordability measures to capture the full scope of costs being passed onto households.

Recognizing the full costs to households accessing safe water.

Indirect and non-monetary costs appear critical to understanding affordability in some contexts. While Hutton 2012 proposes an indicator inclusive of all financial and economic costs, only a few studies measure coping costs in addition to the direct costs of water in their analysis (Moore *et al* 2011, Amit and Sasidharan 2019). A similar observation was made in a review of water insecurity measures by Wutich et al. 2017, who identified metrics of non-market entitlements and costs as underdeveloped compared with water affordability measures (Wutich *et al* 2017). This has implications for the way affordability is represented and for the way we interpret results regarding affordability's association with other dimensions of water access.

Water affordability research needs to better define the full scope of direct and indirect costs relevant to households and communities in the U.S. Specifically, research into multiple water sources (i.e. public water supply, private wells, bottled water) and their attending costs could help prioritize which indirect costs need to be measured and/or monetized. We found the lack of affordability research on domestic well owners in the US surprising given that an estimated 14% of the US (44.5 million residents) rely on domestic wells that are not regulated or monitored. Several studies have shown disparate exposures to microbiological and chemical contaminants among well-owners in the US (Heaney *et al* 2011, 2013, Stillo and Gibson 2017, Murray *et al* 2018, Gibson and Pieper 2017). It is likely that households using private wells and septic systems face greater time, effort, and treatment costs to maintain safe drinking water and sanitation needs.

Where water quality is known to be poor (e.g. through comparison with an area's violations with federal or state standards), affordability studies in the U.S. could be augmented by estimating household costs for treatment, collection, and storage and/or the use of bottled water. These costs could be incorporated into income burden measures through household surveys or as a sensitivity analysis to explore the upper range of costs for households in aggregate studies of water affordability.

More theoretical and case-study work on household level affordability required to justify complex metrics in practice.

Households with very few resources may be forced to reduce spending on water overall ("under-consuming" (Gawel *et al* 2013)) or reduce spending on other essential needs, like health care spending or food (Cory and Taylor 2017). More theoretical and empirical assessments are needed to understand how affordability impacts household's ability-to-pay and their spending, broadly. A key feature of water affordability is that it does not come at the cost of other essential needs. It is unclear if *any* of the metrics reviewed can capture this phenomena.

The two measures that seek to indicate this are the Residual Income Approach (RIA) and AR_{20}. The broad assumption in both these measures is that after we account for expenses toward other essential needs, we can estimate the affordability of water relative to discretionary income (which includes spending on things like transportation, clothing, reading, etc). In the original paper developing the AR_{20} metric, Davis and Teodoro state that AR_{20} "reflects economic tradeoffs that customers face due to the costs of basic water and wastewater services" (Davis and Teodoro 2014). It is unclear however how water bills at 10% of discretionary income reflect whether households do not indeed need to trade-off with other expenses, nor how 10% of discretionary income relates to normative criteria of poverty or reasonableness of burden.

More research is needed, but some evidence suggests that evaluating water bills as a ratio of discretionary income assumes more about the internal structure of household budgets than evidence supports. Cory and Taylor (2017) model the internal structure of spending budgets over time in the U.S. Consumer Expenditure Survey for different income levels and evaluate how water bill increases effect spending. They demonstrate that households respond to an increase in the cost of water by distributing cuts in their budget not only in discretionary income, but also on spending for health care and in some cases, food (Cory and Taylor 2017). This has some implications to how far we can interpret AR_{20} as representing a trade-off. If water and sewer bills are less than 10% of discretionary income they are considered affordable, but it could well be that households are decreasing spending on other essential needs, regardless.

This does not negate the value of AR_{20} and RIA as tools in the toolbox by any means. Indeed the assumption about budgets is not entirely unique to these measures, but as they are more developed in AR_{20} and RIA metrics it is worth trying to understand if measures that require more assumptions and data are moving closer to measuring the trade-off that they seek to capture (between water and other essential needs). Research is needed to clarify the range of essential needs and their costs to households, and how this impacts affordability measures. This is no trivial task–as defining essential needs beyond water and estimating their costs is bound to introduce substantial uncertainty given current data availability.

Prioritize equity analyses with attention to historically marginalized groups.

While a comprehensive treatment of various equity concepts is beyond the scope of this review (See: Bakker 2001, Davis 2005), the literature demonstrates several approaches, and we provide recommendations on areas for growth. The human right to water defines equity for water access in distributional terms (with affordability as "ability to pay" being a core feature). The normative principle of non-discrimination further extends the social equity concept to include historical injustices.

Non-discrimination and equity concerns can be incorporated into affordability measures through disaggregated analysis of any and all measures along lines of race/ethnicity, income, and in conjunction with other indicators of household or community vulnerability. Future studies should prioritize affordability impacts across race/ethnicity given the historical marginalization and inequities faced by communities of color in U.S. water access challenges (Wescoat *et al* 2007, Balazs and Ray 2014, Cory and Taylor 2017).

New analyses to unpack the role of ecological sustainability in household affordability.

The relevance and focus on ecological sustainability in the context of household water affordability is not clearly specified in the literature, but several directions are promising for expanded work in this area. First, our review identifies the focus on essential needs water use as important for aligning social and ecological goals. Basic needs volumes should be a focus for affordability measures, but applied with critical attention to the limitations of this approach (Gawel and Bretschneider 2016, Howard and Bartram 2003). Where possible, studies could use a range of essential use volumes, conduct sensitivity analyses, or estimate water needs by household size to avoid compromising vulnerable groups. These include households with many people or households with sick or pregnant people. Another analysis to support this approach could be to assess the extent to which the first block in an increasing rate structure or subsidies are affordable and adequately provide essential needs water (Martins *et al* 2013, Molinos-Senante and Donoso 2016).

Secondly, the efficiency of on-premise pipes, fixtures and appliances are all important factors influencing water consumption. Careful selection of basic water requirements or estimates of water use across household water infrastructure quality types could improve representation of various sustainability and equity challenges simultaneously.

Finally, the precarity of freshwater resources in a changing climate is a pressing challenge to water access and water quality. Mack and Wrase (2017) and Frankhauser and Tepic (2007) measure how affordability might change with 'rate shocks', which could serve as a proxy for the impacts of environmental change on affordability (Mack and Wrase 2017, Fankhauser and Tepic 2007).

Examine distributions, use multiple measures, and holistically assess water access.

Research is unlikely to unearth a threshold that definitively captures water affordability or poverty (in the case of the residual income approach). Implementing affordability measures and demonstrating the distribution of results alongside comparison to several thresholds may provide some benchmarking points without forcing a binary delineation of affordability onto household and communities. Research should develop to identify multiple types of affordability criteria in order to expand the representation of affordability challenges as relate to the key features of affordability as a human right and sustainable development goal. At the same time, what truly counts as affordability is a question for local and community determination and it is unlikely to be determined outside of local contexts.

Finally, affordability should ultimately be measured and discussed in relation to other indicators of physical access, water quality, and non-discrimination (Luh *et al* 2013, Kayser *et al* 2013, WHO/UNICEF JMP 2015, Feinstein 2018, Balazs *et al* 2019, Wutich *et al* 2017). To better understand how water affordability hinders or enables the equitable provision of water, however, its measurement needs clear specification. There is not one way to measure water affordability, just as there is not one way that people access water or bear the burden of its costs. Nonetheless, our review identified several areas for researchers and practitioners to consider when developing water affordability metrics. We use the definitions and normative aims of the human right to water and SDGs to elucidate the conceptual scaffolding of water affordability and what it “should” measure. Transparency in what choices analysts make will improve comparability across studies and our knowledge of affordability challenges overall.

CHAPTER 3

3. WATER AFFORDABILITY AND HUMAN RIGHT TO WATER IMPLICATIONS IN CALIFORNIA[2]

Abstract

Water affordability is central to water access but remains a challenge to measure. California enshrined the human right to safe and affordable water in 2012 but the question remains: how should water affordability be measured across the state? This paper contributes to this question in three steps. First, we identify five key dimensions of robust water affordability measures (scale, the minimum volume of water needed to meet 'basic' needs, available income, and criteria for affordability) and two cross cutting themes (social equity, sustainability). Second, using these dimensions, we develop three affordability ratios measured at the water system scale for households with median, poverty level, and deep poverty (i.e., half the poverty level) incomes. We also estimate the percentage of households in each system below poverty and deep poverty levels. Third, we analyze our results disaggregated by a key characteristic of water system vulnerability in California – water system size. Using multiple measures conveys a fuller picture of affordability challenges given the known limitations of specific affordability indicators. Our results demonstrate the extent to which water is, on average, more affordable for households at the median income level than for households at the county poverty and deep poverty levels. We identify several water unaffordability scenarios that have different policy implications, such as very small systems with high water bills and low-income households within large community water systems. This paper presents a systematic approach to creating affordability ratios, and represents the first statewide assessment of water affordability in California's community water systems.

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2 This chapter has been submitted to a journal in edited form and has been approved for use in this dissertation by my co-authors, Carolina Balazs and Isha Ray.

3.1 Introduction

Water affordability is central to water access but remains a challenge to measure. The United Nations General Comment No. 15 (GC15) on the human right to water defines water as economically accessible if the direct and indirect costs associated with water and sanitation do not impact a person's access to other essential rights (e.g., food or shelter) (United Nations Committee on Economic Social and Cultural Rights 2002, UN 2005). Following GC15, water affordability became established as a conceptual pillar of human right to water frameworks (United Nations Committee on Economic Social and Cultural Rights 2002, Roaf *et al* 2005, Feinstein 2018, Villumsen and Jensen 2014). The United Nations' Sustainable Development Goal (SDG) report followed suit with the target to "achieve universal and equitable access to safe and affordable drinking water for all" by 2030 (United Nations 2018). The SDG indicators have not identified an approach to measure water affordability, however, due in part to inadequate data. Thus there is a gap between the aspirations for affordable water for all and the availability of measures to track this aim; this gap is generally recognized in the literature (Gawel and Bretschneider 2016, NAPA 2017). How can states leverage existing data to define and monitor water affordability at scales relevant to policy-making? Answering this question is necessary to support efforts to meet the SDGs and the human right to water across the globe. This chapter proposes a new approach for developing water affordability measures and demonstrates its use in California community water systems.

International work dominates safe water policy and research agendas (Smets 2009, Hutton 2012), but access to clean and affordable drinking water is a growing challenge in the United States (Wescoat *et al* 2007, Allaire *et al* 2018). The severity of water access inequities in the U.S. jarred national consciousness with the uncovering of Flint's lead crisis (Pieper *et al* 2017) and of thousands of water shut-offs in Baltimore and Detroit due to unpaid bills (OHCR 2014). Despite the growth in studies evaluating affordability, there is an active debate on how to measure it (NAPA 2017, National Drinking Water Advisory Council 2003, U.S. Conference of Mayors 2014). Scholars from geography, public policy, and microeconomics alike measure the cost of drinking water relative to income. This ratio, compared against a specified threshold, identifies water (un)affordability (García-Valiñas et al., 2010; Hutton, 2012; Jepson, 2014; Mack & Wrase, 2017; Smets, 2009; Teodoro, 2018). Several case studies in the U.S. highlight water affordability challenges in California (Christian-Smith *et al* 2013, Pierce and McCann 2015, Hanak *et al* 2014, U.S. Conference of Mayors 2014), Michigan (Rockowitz *et al* 2018), the US-Mexico border in Texas (Jepson 2014), and the U.S. overall (Mack and Wrase 2017, Teodoro 2018). Despite differences in scale, geography, and methods, most of these studies describe a similar story: lower-income households or communities, even in relatively rich contexts, frequently face water affordability challenges.

Concerns over how affordability measures should incorporate economically vulnerable groups are paramount, but additional debates include questions of scale, the volume of water use that should be measured, and which criteria should be used to judge affordability. In this paper, we contribute to the emerging discussion on the meaning of drinking water affordability and the tractability of its measurement within a human right to water framework. Our study site is California, and our focus is the state's 2,901 community water systems (CWS). CWSs are public water systems that serve water year-round to at least 25 people or have 15 or more service connections (Health & Safety Code 2017). We ask: Do Californians served by community water systems face water affordability problems? How can we measure this? To balance analytical rigor with practical usability, we co-produced our research design and resulting measures with California Environmental Protection Agency's (EPA) Office of Environmental Health Hazard Assessment, which is developing approaches to evaluate the human right in California (Balazs *et al* 2019).

We aim to understand the state of water affordability for domestic water used for drinking, hygiene, and cooking in California's CWSs. We do not include wastewater or sanitation costs, due primarily to data constraints. We identify key dimensions relevant to water affordability that are grounded in theoretical and practical considerations (Section 3.2). We then apply these dimensions as a framework to develop a set of affordability measures and household poverty indices (Section 3.3). This allows us to situate our evaluation of affordability ratios within existing public debates and critiques of what affordability 'should' measure (Wutich *et al* 2017). We focus our analysis on system-level metrics to capture household water bill burdens at three income levels. This is the most granular level of data available. Additionally, a water system scale approach enables us to focus on small systems, where population counts are small but the challenges in securing safe water access are big (McFarlane et al. 2019). Finally, we analyze our affordability results by system size. Water system size is measured by the population served and broken into four categories from very small (25-500 people) to large (10,000+ people). Evaluating our data by system size rather than population weighting our results allows us to treat all populations served equally (regardless of the size of their system). To our knowledge, this study is the first state-wide investigation on water affordability for households in community water systems.

We select California as a case study for several reasons. First, water bills in California increased by 42 to 47 percent over the previous two decades, disproportionately impacting households dependent on smaller systems (State Water Resources Control Board, 2015). This trend reflects the broader trend in the U.S.: water prices are rising faster than inflation in urban areas, signaling that affordability challenges will likely grow (Hanak *et al* 2014). Water costs present a considerable burden for lower income communities in rural (Moore *et al* 2011, Christian-Smith *et al* 2013) and urban (Pierce and McCann 2015) contexts, measured at both the household and water system scales. Secondly, water affordability is deeply entwined with inequitable contaminant exposure and broader access challenges. Unaffordable water bills compound and perpetuate water quality problems, leading to a "joint burden" on households and systems (Balazs and Ray 2014). Disparities in drinking water access persist because of inequities in infrastructure (Vanderslice 2011), enforcement and regulatory design (Katner *et al* 2016, Cory and Taylor 2017, Cory and Rahman 2009), uneven monitoring (Heaney *et al* 2011), high tap water and bottled water costs paired with low ability-to-pay (Pannu 2012, Moore *et al* 2011), and low technical, managerial and financial capacity (Balazs and Ray 2014). The impact of unaffordable water on these co-occurring and multilevel burdens is substantial for economically vulnerable households, who may compromise health and food-related expenses to pay for water and utilities (Cory and Taylor 2017).

The opportunities to assess affordability are also considerable in the state. In 2012, California was the first U.S. state to establish *safe, clean, affordable, and accessible* water as a human right (Assembly Bill 685), which became California Water Code the following year (Eng 2012). Despite this bold political action, however, the bill's language does not outline a process to assess and track the human right to water. State efforts to support water systems in alleviating quality or affordability challenges have primarily focused on small systems that are socioeconomically disadvantaged (State Water Resources Control Board 2018). Currently, an estimated 20% of eligible households actually receive subsidies, but the State Water Resources Control is investigating policy options for household-level bill assistance (State Water Resources Control Board 2019b). Grassroots organizations, water suppliers, and communities are actively debating the meaning of water affordability and programs to address it (c.f. public comments on a low-income rate assistance program). This chapter contributes to this discussion by offering an approach to measure water affordability for California that supports monitoring efforts towards the realization of the human right to water.

3.2 Core Dimensions for Constructing Affordability Ratios

As reviewed in Chapter 2, international and U.S. water affordability indicators for human right to water monitoring rely heavily on ratio-based approaches (Hutton 2012), which calculate water costs as a proportion of income, either at household or aggregate scales such as a utility (Teodoro 2018). Most studies do not explicitly outline the reasons for choosing a particular affordability metric or emphasize what the resulting measures do or do not represent in a comparable way across studies. To promote transparency and consistency across affordability assessments, we develop our metrics in conversation with the themes of water affordability emergent from the review in Chapter 2. Below, we briefly summarize core dimensions relevant to measuring affordability ratios, discuss the critiques associated with each dimension, and identify approaches to address the identified limitation(s). Specifically, we focus on affordability ratio measures.

Spatial and temporal scale and scope. Water affordability is experienced at the household scale (Martins *et al* 2016, Jepson and Vandewalle 2016, Gawel *et al* 2013, Vanhille *et al* 2018), but policy measures typically use aggregated data at scales ranging from water systems (Teodoro *et al* 2018) to nation states (Smets 2017). Most research advocates disaggregating results by income groups or household types to better capture the heterogeneity of affordability challenges that occur within larger geographies. Evaluating water affordability over time or evaluating future rates (Fankhauser and Tepic 2007, Mack and Wrase 2017) is less common, but are important to identifying e temporal dynamics that cross-sectional studies do not capture.

Water to meet basic needs. Studies that evaluate affordability based on average household water use risk under- or over-estimating affordability problems because they do not focus on essential volumes of water (Gawel *et al* 2013). Human right to water efforts are concerned with filling basic needs, as opposed to luxury water uses (e.g., filling pools). To determine affordability of the volume of water essential for meeting basic needs, it is necessary to decide what constitutes this minimum volume. The literature offers several approaches to this question: 1) determining basic needs water from demand functions (García-Valiñas et al., 2010), 2) estimating minimum basic water requirements for universal norm-setting (Gleick 1996, Howard and Bartram 2003), and 3) deriving location-specific estimates of water use based on ‘reference budgets’–or the minimum amounts of water needed for washing, cooking, hygiene, and consumption (Vanhille *et al* 2018, Feinstein 2018).

Counting all the costs. Affordability ratios ideally include the cost of water, sewer, and any indirect or coping costs that households incur. For example, a study may include replacement costs (such as purchasing bottled water for drinking) on top of the water bill in an attempt to better capture costs for safe, basic needs water (Moore *et al* 2011).

Available income. The denominator of an affordability ratio implicitly defines the amount of money a household has available to spend on water. In the absence of household level income and expenditure data, studies use proxy measures like gross income or expenditures to capture available income. For low-income households, expenditures may be a more accurate measure of available income than reported income due to the nature of lower-income jobs (Deaton 1997). The limitations of using gross income in affordability ratios can be addressed by evaluating water affordability as a proportion of disposable income (Gawel *et al* 2013, Smets 2009), or disposable income less modeled estimates of essential expenditures, which results in an estimate of discretionary income (Teodoro 2018).

Criteria for affordability. Thresholds to determine whether water is unaffordable range from water bills being 1.5% to 5% of income, varying as to whether they include both drinking water and sanitation, as outlined in Table 2.1 (State Water Resources Control Board 2018, US EPA 1998a, Hutton 2012, Banerjee and Morella 2011). EPA’s affordability frameworks for drinking water (1998) and wastewater (1997) discuss household affordability as a component of a water system’s financial capacity, largely driven by cost-recovery concerns (US EPA 1997, 1998a). Recent recommendations to EPA’s frameworks argue for increased focus on affordability for low-income households and on the prevalence of poverty in water systems (NAPA 2017).

Most states use lower thresholds in their assessments of affordability compared with EPA’s 2.5% threshold for median income levels. In California, the State Water Resources Control Board has used a threshold for water bills as 1.5% of median household income to provide financial aid to lower-income water systems. The most frequently cited threshold for water and wastewater bills as a percentage of disposable income is 3% (United Nations Development Program 2006, Reynaud 2006), and recently 5% was proposed to evaluate drinking water as a percentage of disposable income less essential expenditures (Teodoro 2018). Some researchers (Sawkins and Dickie 2005, Balazs *et al* 2019) present the distribution of results disaggregated by multiple categories (e.g., system size; income decile) and interpret affordability as a spectrum from more to less affordable, without normatively declaring water to be ‘affordable’ or ‘unaffordable’. This may be appropriate for tracking and monitoring progress of water as a human right over time.

Social equity and non-discrimination. Affordability in the human right to water context demands attention to economically vulnerable households and other historically marginalized communities. The U.S. EPA framework for water affordability has led to critique for focusing only on median income levels to represent affordability, thereby underrepresenting economically vulnerable groups (Environmental Law Clinic, 2017; U.S. EPA Environmental Financial Advisory Board, 2014). Options to address this critique include: 1) estimating affordability for lower-income households (Reynaud 2006, Gawel *et al* 2013, Teodoro 2018) and across income groups (Martins *et al* 2016, Vanhille *et al* 2018, Sawkins and Dickie 2005), and 2) with caveats, using affordability ratios that rely on median income levels in very low-income communities.

Ecological sustainability. The relevance of ecological sustainability in measuring affordability is somewhat challenging to incorporate into measures, but may be relevant to consider in achieving broader sustainability goals. Measuring affordability for basic needs ensures that affordability is not measuring excessive water use (e.g. pools), but this needs to be applied with care because different households require different amounts of water (Gawel and Bretschneider 2016, Howard and Bartram 2003). Other approaches include measuring affordability in response to rate shocks from environmental events (e.g. drought).

These core dimensions of affordability ratio measurement can inform the development of new metrics. Consideration of these dimensions requires linking conceptual ideas about what affordability ratios capture with the data limitations inevitable in any study. This rubric thus has the potential to highlight opportunities to improve the process of affordability ratio design while ensuring that resultant measures are discussed transparently in terms of what they can and cannot do.

3.3 METHODS AND DATA

Our research questions–what is the state of water affordability in California and how can we measure this?–required methodological and empirical developments that we undertook in two stages. First, we designed three affordability ratios and two household poverty indices based on the above dimensions, and given available data. We then evaluated the results by community water system size. Two data challenges are explored in the development of the metrics below: high levels of missing data in our dependent variable (affordability ratios based on water bills and income) and concerns about data reliability. We address missing data by adjusting for measured confounders of missingness and discussing potential bias. Data reliability concerns are addressed through a sensitivity analysis.

3.3.1 *AFFORDABILITY RATIOS*

We developed three affordability ratios (Eqn 1-3) that collectively address several aspects of the core dimensions (indicated in italicized font below). The ratios capture the impact of water bills on households earning three different income levels (median household income and two poverty-level incomes) using data collected at the water system *scale*. These measures are potential affordability ratios (PAR). The PAR does not indicate actual volumes of water consumed by a household, but rather the minimum theoretical amount that a household should consume to meet its basic needs. For this reason the affordability ratio is labeled as potential (García-Valiñas et al., 2010; Gawel et al., 2013; Kessides et al., 2009; Miniaci et al., 2008). The PAR offers the benefit of 1) assessing water affordability for the *volume of water required to meet essential needs* and 2) providing the basis for comparing the income burden of water bills (for a fixed volume of water) across multiple units of analysis (e.g., households or water systems) over time.

In California, CWSs have the most granular and comprehensive data availability. This is also the scale for water quality and water access indicators for monitoring California's human right to water (Balazs *et al* 2019). We obtained a list of 2,903 CWS that were active in 2015 and their geographic boundaries from the (California Environmental Health Tracking Tool n.d.).

To capture the *direct cost of water* to households, we used water bill data from the California State Water Resources Control Board's Electronic Annual Report (eAR) dataset. The eAR survey asks systems to report: what was the average household water bill for 6, 12, and 24 hundred cubic feet? We use data from 2015 as that was the most recently available data at the time of the study. The average water bill includes fixed and variable costs of water to households, but sanitation costs and indirect costs (e.g., treating water at home or bottled water use) are not included or estimated. We evaluate affordability for the minimum volume of *water to meet basic needs* by using water bills reported by systems for 6 HCF. This volume is equivalent to 4,488 gallons per month, or approximately 37 gallons per capita per day (gpcd) in a four-person family and 49 gpcd in a three-person family. This range (37-49 gpcd) aligns with California's conservation goals of 55 gpcd (California Water Code §10608.2) and recommendations to evaluate water affordability at 43 gpcd (Feinstein 2018).

Water bill data were cleaned using R Version 3.5.1 (R Core Team 2018) and adjusted, as needed, in accordance with predetermined data cleaning criteria (Appendix B1). Given the absence of a prior state-wide assessment of eAR survey accuracy and a high level of missing data, we took several steps to address measured confounders of missingness and explore the sensitivity of our results (described in below and Appendix B3-B5). First, systems were flagged as missing if they lacked data. Secondly, we used an adjusted box plot for skewed distributions to identify potential outliers (Hubert and Vandervieren 2008); this approach yielded upper and lower limits on water bills based on their distribution. Water systems whose water bills were below or above the fence were flagged for reliability and contacted by phone to verify the accuracy of the potential outliers and contextualize the sensitivity analysis (Appendix B3).

3.3.2 *Affordability Ratio for Households Earning Median Household Income*

The first affordability ratio evaluates water bills for 6 HCF for households earning at the median household income (MHI) of a water system's income distribution (AR_{MHI}). High values of AR_{MHI} can signal that water affordability is a problem for a majority of households within a water system. We calculated the ratio as in Equation 1:

$$AR_{MHI} = \frac{Monthly\ Water\ Bill\ at\ 6\ HCF * 12\ months/year}{Annual\ Median\ Household\ Income\ in\ Water\ System} \times 100 \qquad (1)$$

We estimated an MHI for each water system using block group level MHI data from the American Community Survey (5-year estimates; ACS 2011-2015). Because community water systems do not share boundaries with census-designated geographies, we aerially apportion census block group data to water system boundaries using an areal-household weighting method (OEHHA 2017a). We first assign the number of populated households in census blocks by the proportion of block area overlapping with water system boundaries in ArcGIS. The number of households served by a water system by block are summed to their block group ID and used to estimate the number of households served by a water system within each block group. Using these aerially apportioned household estimates and MHI data by block group, we calculated a weighted average MHI for each water system (See Appendix B2). A limitation of this approach is that small water systems falling within one block group are assigned the MHI of the block group because no data exists to capture inter-block group heterogeneity. Moreover, data within block groups is assumed to be homogenous across the block group.

Systems were flagged as missing data if they had no MHI data and systems missing MHI data for more than 15% of the households (see Appendix B4). We also assessed census data reliability for systems falling within one block group using census-derived coefficients of variation. Systems were flagged for removal in a sensitivity analysis if the census MHI estimate: 1) had a coefficient of variation greater than 50 *and* 2) the standard error of the estimate was greater than the mean standard error of all California block groups for the estimate (OEHHA, 2017).

3.3.3 *Affordability Ratios for Households Earning Poverty and Deep Poverty Levels*

AR_{MHI} cannot indicate water bill impacts on *economically vulnerable households* unless a majority of households in the water system are low-income. We therefore developed affordability ratios for two types of low-income households: those earning at California county poverty levels (Eqn 2: AR_{CP}), and those in 'deep poverty' for their counties (Eqn 3: AR_{DP}):

$$AR_{CP} = \frac{Monthly\ Water\ Bill\ at\ 6\ HCF * 12\ months/year}{County\ Poverty\ Level\ for\ Water\ System} \times 100 \tag{2}$$

$$AR_{DP} = \frac{Monthly\ Water\ Bill\ at\ 6\ HCF * 12\ months/year}{\frac{1}{2}(County\ Poverty\ Level\ for\ Water\ System)} \times 100 \tag{3}$$

The county poverty level (CP) reflects essential household expenses, or a minimum disposable income, adequate for a household of four to stay out of poverty within their county. Deep poverty (DP) is defined as 50% of CP to capture extreme economic vulnerability. While perfectly correlated with county poverty, deep poverty provides a snapshot of the most economically vulnerable households. Both measures adjust for key differences in expenses across counties, such as housing costs (Bohn *et al* 2013b). We acquired county poverty data from the Public Policy Institute of California and assigned every water system the poverty and deep poverty level of its respective county. Of California's 58 counties, 38 have unique poverty levels and the remaining 20 counties are divided into three groups with equal poverty levels (due to census data suppression criteria).

Affordability ratios for households earning at the county and deep poverty levels improve the representation of *available income* in the denominator by approximating a minimum disposable income for economically vulnerable households. Our measures do not remove essential, non-water expenditures from the denominator, as this data is not available. As such, when AR_{CP} and AR_{DP} are very low (i.e., water is relatively affordable), we cannot identify if households are compromising other essential needs.

3.3.4 *CRITERIA FOR AFFORDABILITY*

We choose not to use affordability thresholds as binary arbiters of (un)affordability and thus we do not determine explicit *criteria for affordability*. We do, however, compare our results with existing thresholds (e.g., 1.5% of MHI and 3% of disposable income) for illustrative purposes, recognizing that they may not be perfectly comparable thresholds because of data choices for developing the affordability ratios in this study. We interpret affordability on a spectrum from more affordable (i.e., lower values of each ratio) to less affordable. Additionally, we analyze how the distribution of results changes when we look at affordability ratios across different system characteristics.

3.3.5 *HOUSEHOLD POVERTY INDICES*

AR_{CP} and AR_{DP} do not indicate the prevalence of low-income households facing these affordability ratios within a water system. To capture the number of households facing at least AR_{CP} or AR_{DP}, we summed the number of households within a water system at or below the county poverty level (HH_{CP}) and deep poverty level (HH_{DP}). We then divided these sums by the total number of households in each water system to calculate percentages, resulting in two household poverty indices (Eqn 4-5):

$$HH_{CP} = \frac{\sum Households\ in\ Water\ Sytsem \leq County\ Poverty\ Level}{Total\ Households\ in\ Water\ System} \times 100 \tag{4}$$

$$HH_{DP} = \frac{\sum Households\ in\ Water\ Sytsem \leq County\ Deep\ Poverty\ Level}{Total\ Households\ in\ Water\ System} \times 100 \tag{5}$$

To estimate HH_{CP} and HH_{DP}, we used census block group household count estimates of: 1) total number of households and 2) the number of households within each of sixteen income levels. We applied the areal-household weighting method described above to obtain water-system level household count estimates (Appendix B2). Given that the Census bins the number of households into discrete income levels (e.g., \$15,000-\$25,000), we used linear interpolation in R Version 3.5.1 to sum the number of households within each water system falling at or below the county poverty and deep poverty levels (R Core Team 2018).

We applied the same sensitivity analysis described for MHI (flag for missingness = missing or partial data; flag for reliability = census reliability) to all census household count estimates for systems falling within one block group. However, census reliability exclusion criteria for household count estimates by income level differed slightly from the exclusion criteria for MHI. Systems were flagged as unreliable for sensitivity analysis in our assessment of household poverty indices if twenty percent of the underlying household count estimates across income levels were unreliable for a system (Appendix B4).

3.3.6 *Core dimensions for Developing Affordability Ratios*

Table 3.1 summarizes how the three affordability ratios and two household poverty indices collectively address the seven key dimensions associated with affordability ratios outlined in Chapter 2 and summarized above.

Table 3.1 Summary of key dimensions to address for the development of affordability ratios.

Plain text summarizes how the ratio used in this paper addresses the dimension. Text in italics indicates aspects of affordability *not* captured by these measures (i.e., their limitations).

Key Dimension	*Median Affordability Ratio (AR_{MHI})*	*Poverty Affordability Ratio (AR_{CP})*	*Deep Poverty Affordability Ratio (AR_{DP})*
Spatial and Temporal Scale and Scope	Water system scale aligns with current human right to water efforts in California (Balazs *et al* 2019) and is the scale with the most comprehensive, state-wide data. *Affordability is approximated for three household-level incomes, but ratios are not evaluated for each household, as this would require a micro-level study.* *Affordability analysis is cross-sectional, but metrics are part of a human right to water monitoring effort and will be measured over time* (Balazs *et al* 2019).		
Water for basic needs	Water bill evaluated at 6 HCF per month to approximate the volume of water needed to meet basic needs for households and to parallel California conservation goals. *Variations in basic needs for vulnerable groups (e.g. families with babies or medical needs), larger or smaller household sizes, and/or differing geographies are not addressed.*		
Costs	Use of reported average water bills for 6 HCF per month, which includes the price of water and any fees or subsidies included by the water system. *Sewer costs and other costs like bottled are not included due to limited data. If households obtain basic needs water from alternate sources (e.g. bottled water), these sources are not reflected in water bills.*		
Available income	*Median household incomes do not capture the heterogeneity of incomes within water system.* *Median household incomes are gross income levels and do not remove non-essential or non-water essential expenditures from income.*	County poverty levels incorporate cost of living and minimum essential needs budget. County poverty levels approximate disposable income at poverty level. *County poverty levels do not subtract non-water essential expenditures from income.*	Deep poverty levels reflect households with an extreme income constraint, at *half* disposable income for poverty level households. *Deep poverty levels do not subtract non-water essential expenditures from income.*
Criteria for affordability	Binary affordability criteria are not used. Instead, this study presents distribution of data and highlights a range of scenarios and analyses based on key water system characteristics. Metrics are part of a broader human right to water effort that will analyze trends over time in California.		
Social equity & non-discrimination	Income at 50th percentile of low-income system can yield indicate concentrations of low-income households. *Social equity is not explicitly addressed.*	Social equity is partially addressed through a focus on economic vulnerability, which is explicitly addressed by focusing on households earning at the county poverty level, an income level that approximates disposable income for vulnerable households.	Social equity is partially addressed through a focus on economic vulnerability, which is explicitly addressed by focusing on households at the deep poverty level, an income level that approximates disposable income for the most vulnerable households.
Ecological sustainability	Focus on essential needs water is in line with potential affordability ratios that do not capture luxury uses of water. *Added fees due to drought could be included in a water system's reported bill but this is not disaggregated.*		

3.4 ANALYSIS

3.4.1 *EVALUATION OF RESULTS BY WATER SYSTEM SIZE*

Our results include descriptive statistics of the data inputs and outputs for the measures described above, including water bills, affordability ratios (AR_{MHI}, AR_{CP}, AR_{DP}), and the percentage of households below poverty levels (HH_{CP}, HH_{DP}) in California. We assess our results by system size. Systems are considered very small if they serve 25-500 people, small if they serve 501-3,300 people, medium if 3,301-10,000 people, large if 10,001-100,000 people, and very large if they serve >100,000 people (US EPA 2018). We collapsed the very large and large system size category (10,000+ people) for the statistical analysis to improve the balance of system numbers across groups. System size is a common delineator of a water system's technical, financial, and managerial capacity (State Water Resources Control Board 2018).

3.4.2 *MISSING DATA*

We investigated measured confounders of missingness in the affordability data to reduce bias in our study caused by missing data. We did this in three steps. We stratified the community water system list by the four system size categories; we modeled missingness (i.e. if a system missing water bill or income data) within each size category using variables we thought might be correlated with affordability; and then we investigated marginal effects of these variables on whether a system was missing data within each size category. To do this we coded systems with a 1 if they were missing water bill or income data and a 0 if they had data. We ran 7 separate logistic regressions within each of the 4 size categories.

Potential confounders of missingness included water source type (i.e. surface water or groundwater), water system ownership type (e.g. private or public), and water board governance region from the Safe Drinking Water Information System (SDWIS). Other potential confounders included percent renters, percent households under two times the poverty level, and percent people of color (all non-white race/ethnicity categories combined) from the American Community Survey (ACS) 5-Year averages (2012-2016). ACS variables were aerially apportioned to water systems following the same method described for median household income, but using population instead of household weighting. When one of the potential confounders was found to significantly increase the odds of a system having missing data at alpha = 0.05, we incorporated the variable as a covariate in the analysis of affordability ratios by system size. The conceptual reasoning here to reduce confounding due to missingness when analyzing the affordability ratios. Covariates that are highly correlated with the outcome variable (affordability ratios) and missingness can potentially reduce bias and variance in our final model.

3.4.3 *STATISTICAL ANALYSIS*

We estimated the association of affordability ratios (AR_{MHI}, AR_{CP}, AR_{DP}) and system size (very small, small, intermediate, large) using three separate generalized linear regression models (one for each ratio) in R Version 3.5.1. We compared a model of affordability ratios regressed against several continuous and categorical variables identified as confounders of missingness first with and then without the system size variable. Low Akaike information criterion (AIC) and a significant F test (alpha =

0.05) comparing the two models served as an omnibus test for the influence of size on affordability ratios. We then estimated adjusted mean affordability ratios and 95% confidence intervals by system size (using the *ggeffects* package in R) and conducted Tukey's post-hoc tests adjusting for multiple hypothesis testing using the *multcomp* package (Hothorn *et al* 2014).

As we were able to estimate household poverty indices (HH_{CP} and HH_{DP}) for the entire water system list with geographic boundaries (n=2,882), we performed One-way ANOVA regressing each poverty index against water system size for the full system list (n=2,882) and for the sample list (n=1,501). We used Welch's test to account for unbalanced groups and unequal variances among groups with the *userfriendlyscience* package in R Version 3.5.1 (Peters 2017). Where Welch's ANOVA was significant, we conducted Games-Howell post-hoc tests to evaluate difference of means for household poverty indices across system sizes (Hothorn *et al* 2014).

We assessed normality and variance of all results–AR_{MHI}, AR_{CP}, AR_{DP}, HH_{CP} and HH_{DP}–for the full distribution and by system size. We log transformed all affordability ratios and square-root transformed HH_{CP} and HH_{DP} to account for non-normality. Residuals from models were evaluated for normality using Shapiro-Wilk's tests. Shapiro-Wilks tests are highly sensitive to deviations from normality and QQPlots were consulted given the large sample size (McDonald 2014).

3.4.4 *Sensitivity Analysis*

As described above, we identified potentially unreliable water bill data and unreliable census income data. We ran models and analyses for water system affordability ratios and poverty indices by system size with all systems that have data, and then excluding systems that have potentially unreliable affordability data. This allowed us to evaluate the sensitivity of our results to data quality concerns.

3.5 Results

3.5.1 *Final Study List*

2,901 California community water systems were active in 2015. Of the 2,901 active community water systems, 2,882 systems have water system boundary data; the 19 systems missing boundary data are primarily prisons which did not charge or report water bills. Of the 2,882 remaining systems, 1,501 systems had water bill data at 6 HCF and income data to assess affordability ratios and supporting household poverty indices (Appendix B1). These systems serve approximately 33.2 million Californians, or 95% of the state's population served by community water systems in 2015.

Due to the high number of systems with missing or inadequate data (n=1,400 out of 2,901), we evaluated potential bias in our sample by comparing the complete case sample to the overall water system list by a variety of water system characteristics, including water system source and social-demographic data (Appendix B6). We find that the sample is relatively representative of the full population of water systems across key water system characteristics, but that it is moderately biased against representation of very small systems and those with very low median household income levels (Table B6). To mitigate bias, we evaluated potential confounders of missingness by system size for inclusion in an adjusted model of affordability ratios and post-hoc tests by water system size.

Statistical analyses were run with and without systems flagged as potentially unreliable to test the robustness of results for affordability ratios and then separately for the household poverty indices. 148 systems were flagged due to the water bill outlier assessment (n = 98), income data missing for more than 15% of households (n = 46), or census reliability exclusion criteria for median household income estimates (n = 8). Four systems fell into more than one of these categories. An additional 227 systems were flagged as having potentially unreliable household count estimates (n=227). Exclusion of unreliable systems based on water bill outliers or income data did not affect the overall trend of results across water system size, however there were differences in post-hoc tests for affordability ratios and household poverty indices, which we discuss below and in Appendix B5.

3.5.2 *DESCRIPTIVE STATISTICS*

The variables for constructing the three affordability ratios and two household poverty indices are water bills, median household income, the county poverty income level, and the county deep poverty income level, which we summarize in Table 3.2. For income levels, we present data for the final study sample alongside data for the full list of water systems with boundaries (n=2,882) to indicate potential bias in the crude summary statistics for the study sample. Of the 1,400 systems with missing data, 1,369 systems had missing water bill data (19 of these systems also had no water system boundary, leading to the n=2,882) and 31 systems had missing income data.

Monthly water bills for systems that reported average bills for 6 HCF span three orders of magnitude across systems, ranging from \$3.06 to \$466.00 per month. The average reported bill across systems is \$51.61 (median = \$40.73). Median household incomes across the state range from \$17,400 to \$250,000, with a slight bias toward higher median household incomes represented in the sample as compared with the full community water system distribution. The range of county poverty levels–i.e., the minimum income needed to remain out of poverty for a family of four–and deep poverty levels are \$23,700 to \$36,200 and \$11,900 to \$18,000, respectively. The sample distribution of poverty incomes is relatively similar to that of the overall community water system list. In total, there are 41 systems (2.7% of 1,501) in the sample with median household incomes below their respective county's poverty level, as compared with 141 systems (5.9% of 2,882) in the full study list. Combined with the higher median household income average in the sample systems, this suggests that our sample has a slight underrepresentation of very low income water systems, which is also identified in Appendix B6.

Table 3.2 Summary statistics for water bills and income data in affordability study sample and full community water system list, for 2015.

	Study Sample (n = 1,501)	All CWSs (n=2,901)
	Monthly Water Bill – 6 HCF ($)	
	n = 1,501	
Average ± SD	52.44 ± 41.35	
Median (IQR)	41.42 (29.19, 61.33)	Full distribution not known
Minimum	3.06	
Maximum	466	
	Median Household Income ($)‡	
	n = 1,501	*n = 2,813[a]*
Average ± SD	64,600 ± 29,200	61,800 ± 27,900
Median (IQR)	58,300 (44,000, 78,400)	55,600 (41,400, 76,400)
Minimum	17,400	13,400
Maximum	250,000	250,000
	County Poverty Level ($)‡	
	n = 1,501	*n = 2,882[b]*
Average ± SD	28,200 ± 3,300	27,800 ± 3,300
Median (IQR)	27,900 (25,100 30,500)	27,000 (25,000 30,300)
Minimum	23,700	23,700
Maximum	36,200	36,200
	County Deep Poverty Level ($)‡	
	n = 1,501	*n = 2,882[b]*
Average ± SD	14,100 ± 1,600	13,900 ± 1,600
Median	14,000 (12,500, 15,200)	13,500 (12,500, 15,200)
Minimum	11,900	11,900
Maximum	18,100	18,100

‡All income data is rounded to the nearest $100 in 2015 dollars.
[a]2,813 systems with available data; 69 water systems had no median household income data available; 19 systems had no spatial boundaries to intersect with census data.
[b]2,882 systems with available data; 19 systems of 2,901 no spatial boundaries to intersect with census data.

Crude descriptive statistics of affordability ratios estimated for the three income levels are summarized in Table 3.3. Across systems in the sample, AR_{MHI} ranges from 0.04% to 13.2%. Households earning median income levels within the sample have relatively low affordability ratios (AR_{MHI}) on average (1.1%). A majority of water systems in this sample have a relatively low AR_{MHI}–the 75th percentile AR_{MHI} is 1.3%. Of the 281 systems with AR_{MHI} greater than California's threshold of 1.5%, 172 systems (or 11% of 1,501 total systems) have median household incomes considered disadvantaged (less than 80% of the statewide median household income). For the 41 water systems with median household incomes lower than the county poverty level, the average affordability ratio is notably higher, at 3.1%.

Nearly a fifth of systems in the sample (19%) have AR_{CP} greater than UNDP's suggested threshold for water and sewer of 3% disposable income (as opposed to thresholds for median household incomes, See Table B8). However, AR_{CP} does not include sewer costs and thus many more systems are likely exceed the 3% threshold. The average household earning at the county poverty level has nearly twice the affordability ratio of the average median household income level (average AR_{CP} = 2.2% and average AR_{MHI} = 1.1%). As the deep poverty level is by definition half of the county poverty level, the average affordability ratio for households at the deep poverty level (AR_{DP}) is double that of AR_{CP}, or 4.5% on average. A quarter of water systems (the 75th percentile) have AR_{CP} and AR_{DP} greater than 2.6% and 5.3%, respectively.

Table 3.3 Crude summary statistics for affordability ratios and household poverty indices measured at the community water system scale (n = 1,501), for 2015.‡

	Affordability Ratio for households – Median Household Income **AR_{MHI} (%)**	Affordability Ratio for households – County Poverty Level **AR_{CP} (%)**	Affordability Ratio for households – Deep Poverty Level **AR_{DP} (%)**	% Households in water system at or below County poverty level* **HH_{CP} (%)**	% Households in water system at or below Deep poverty level* **HH_{DP} (%)**
Average (SD)	1.1 ±1.0	2.2 ± 1.7	4.5 ± 3.5	24 ± 12	10 ± 7
Median (IQR)	0.9 (0.6, 1.3)	1.8 (1.3, 2.7)	3.6 (2.5, 5.3)	23 (15, 31)	9 (5, 13)
90th percentile	2.1	3.9	7.9	40	17
99th percentile	5.1	9.5	19	60	30

‡Estimates are rounded to the nearest tenth of a decimal.

Systems whose water bills are at or below the sample mean water bill of $52.44 have an average AR_{CP} of 1.4%. For households in deep poverty paying at or below $52.44 per month, the average affordability ratio is double this, or 2.8%. These findings indicate that average drinking water bills comprise nearly the entire 3% 'quota' implied by international standards on water and sewer making no more than 3% of a households' disposable income. If we consider the extreme cases, the highest poverty-level affordability ratios are in systems with the highest water bills. Among the 148 systems with AR_{CP} greater than 4% (the top 10th percentile of AR_{CP}), the median water bill for 6 HCF is 2.4 times the state-wide average, or $125.10 per month. This suggests that while poverty may drive low income affordability in many cases, high water bills are also of concern.

3.5.3 *AFFORDABILITY RATIOS BY WATER SYSTEM SIZE*

Of the sample of 1,501 systems, 661 systems (44%) are very small, 304 (20%) are small, 166 (11%) are medium, and 370 (25%) are large. There is a clear gradient in affordability ratios based on water system size (Figure 3.1). Large systems appear to have a tighter and lower distribution of water affordability ratios than very small and small systems across income levels. However, the sample distribution is somewhat biased against representation of systems serving less than 500 people. Of the 2,901 community water systems active in 2015, 62.5% are classified as very small–but 44% of the 1,501 systems in our sample are very small (Table B6). Therefore, very small systems are underrepresented by18.5% in the final sample.

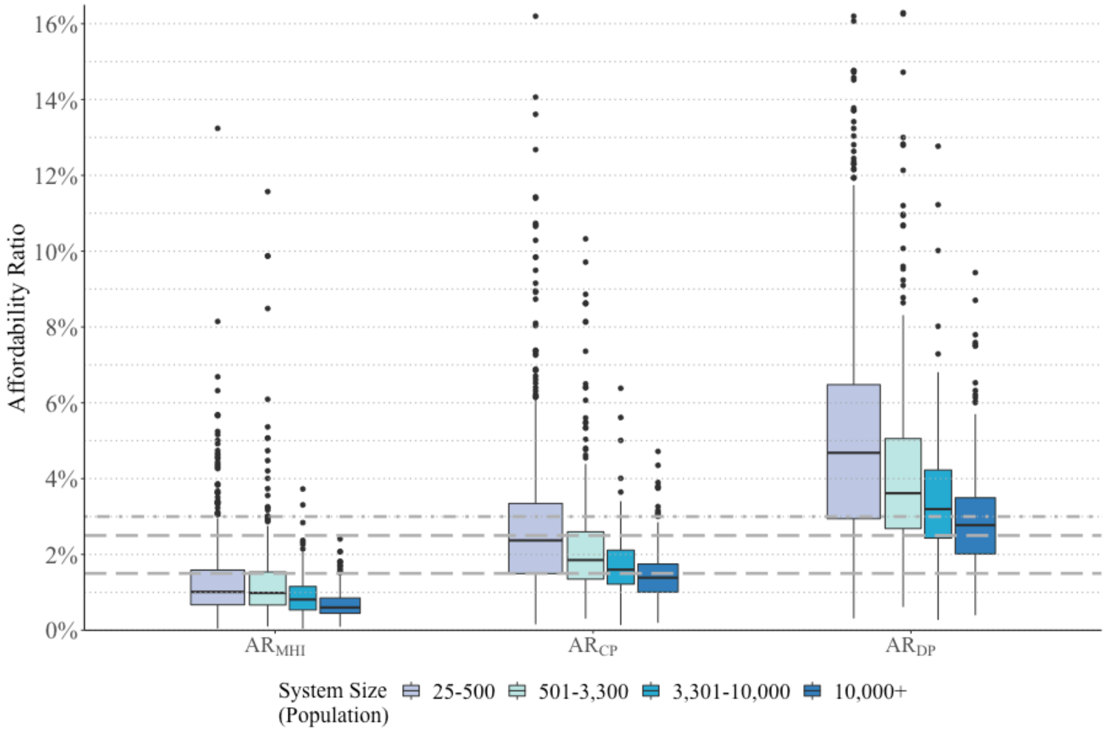

Figure 3.1 Box plots for crude (unadjusted) affordability ratios across income levels, by systems size.

Affordability ratios (AR) measure monthly water bills for 6 hundred cubic feet of water relative to three income levels: AR_{MHI} = median household income level; AR_{CP} = county poverty level; and AR_{DP} = deep poverty level. Long dashed lines represent common thresholds for households earning median household incomes within a water system–1.5% (State Water Resources Control Board 2018) and 2.5% (US EPA 1998a). Dot-dashed lines represent the commonly referenced 3% threshold–which compares water bills (sometimes including sanitation) to income (often disposable income) (United Nations Development Program 2006, Sawkins and Dickie 2005). A newer threshold of 5% of discretionary income for the 20th income percentile has been proposed but includes sewer and so is not shown (Teodoro 2018). As water bills are for a specific volume of domestic water use across ARs, and AR_{CP} and AR_{DP} do not remove essential expenditures from the denominator, the comparison to thresholds is imperfect.

We sought to address this by investigating potential confounders of missingness among system size categories. We found that water system characteristics (ownership, water source type) and region were significantly associated with missingness across the very small size category (measured as an odds ratios with $p < 0.05$) (Table B7). Social-demographic characteristics (percent renters, percent people of color, and percent of households under two times the poverty level) were also significantly associated with missingness, but the magnitude of effects were low (i.e. odds ratios of a system having missing data between 1 and 1.02 for an increase of 1 percent of these social-demographic variables).

We included all of the variables in linear models regressed against each log-transformed affordability ratio (AR_{MHI}, AR_{CP} and AR_{DP}) to minimize bias induced by missing data. A comparison of models with and without system size using AIC criterion and an F-test indicated that the model with covariates and population size improved model fit across all results (Table 3.4). As the focus of this study is on the analysis of affordability ratios by system size, we present estimates and p-values only for water system size category but indicate which variables we adjusted for in the final model. Because AR_{CP}, and AR_{DP} are perfectly correlated in each system–DP is half the CP–we present results for AR_{CP} only. Model fit improves with inclusion of the system size category for both AR_{MHI}, AR_{CP}, and AR_{DP}, and therefore we conducted post-hoc tests to evaluate ratios by water system size.

Table 3.4 General multiple linear regression models for affordability ratios, with and without size categories (n=1,501).

Model log-transformed affordability ratios; coefficients are back-transformed in table.

	AR_{MHI}		**AR_{CP}**	
	Model 1 (without size cateogry)	**Model 2 (with size category)**	**Model 1 (without size cateogry)**	**Model 2 (with size category)**
(Intercept)		0.57 (0.06)***		2.08 (0.06)***
pop bin		*Reference*		*Reference*
25-500		*Reference*		*Reference*
501-3,300		0.92 (0.05)		0.89 (0.04)**
3,301-10,000		0.72 (0.06)***		0.73 (0.06)***
10,000+		0.62 (0.06)***		0.63 (0.05)***
Ownership	**x**	**x**	**x**	**x**
% Under 2x poverty	**x**	**x**	**x**	**x**
% Renters	**x**	**x**	**x**	**x**
% People of Color	**x**	**x**	**x**	**x**
Primary Water Source	**x**	**x**	**x**	**x**
Region	**x**	**x**	**x**	**x**
Observations	1501	1501	1501	1501
R^2 / R^2 adjusted	0.212 / 0.206	0.253 / 0.245	0.120 / 0.112	0.166 / 0.158
AIC	2906.867	2834.371	2765.325	2690.02
F-test	SS = 30.481	F = 26.575***	SS = 30.659	F = 26.732***

Standard error in parentheses, based on log-transformed affordability ratios
* $p < 0.05$; ** $p < 0.01$; *** $p < 0.001$

Table 3.5 summarizes crude and adjusted means and Tukey's post-hoc test results for each affordability ratio. We predicted means by system size adjusting for demographic and water system characteristics to minimize confounding due to high levels of missing data. Adjusted means are lower than observed means across the three affordability ratios. Further, confidence intervals indicate that our sample means are marginally outside the bounds of predicted affordability ratios, but the overall trend of ratios across system size categories remains the same. Very small and small systems have significantly higher average affordability ratios (for all income levels) compared with medium and large systems ($p < 0.001$). Mean differences are significant in pairwise comparisons between very small or small systems (where adjusted mean AR_{CP} are 2.2% and 1.9%, respectively) and medium or large systems (where average AR_{CP} are 1.6% and 1.3%, respectively) ($p < 0.001$).

Table 3.5 Crude and adjusted mean affordability ratios by system size, for 2015.‡

	AR_{MHI}		**AR_{CP}**		**AR_{DP}**	
System size (Number of people) *Number of systems*	**Crude means (SD)**	**Adjusted means (95% CI)**	**Crude means (SD)**	**Adjusted means (95% CI)**	**Crude means (SD)**	**Adjusted means (95% CI)**
Very Small (<500) *n = 661*	1.3 ± 1.1	1.0 (0.96, 1.1)[a]	2.8 ± 2.2	2.2 (2.0, 2.3)[a]	5.6 ± 4.3	4.3 (4.1, 4.5)[a]
Small (501-3,300) *n=304*	1.3 ± 1.3	0.9 (0.9, 1.0)[a]	2.3 ± 1.6	1.9 (1.8, 2.1)[a]	4.5 ± 3.2	3.8 (3.6, 4.1)[a]
Medium (3,301-10,000) *n=166*	0.9 ± 0.6	0.7 (0.7, 0.8)[b]	1.7 ± 0.9	1.6 (1.4, 1.7)[b]	3.4 ± 1.7	3.1 (2.8, 3.4)[b]
Large (10,000+) *n=370*	0.7 ± 0.4	0.6 (0.6, 0.7)[b]	1.5 ± 0.7	1.3 (1.3, 1.5)[b]	2.9 ± 1.4	2.7 (2.5, 2.9)[b]

‡ Results are rounded to the tenth of a decimal for percentages. For adjusted means estimated with the *ggeffects* package in R, covariates were held to average or for factor variables, proportional relative to sample. All data were log transformed for statistical tests and back-transformed for the table. For each measure shown, means that share the same letter column-wise are not significantly different from one another based on Tukey's Honest Difference post-hoc tests on log-transformed affordability ratios in the general linear models shown in Table 3.4. Post-hoc letters were calculated using *multcomp* package in R (Version 3.5.1; R Development Core Team) and ordered to start comparisons with the highest mean value.

Across systems, poverty and deep poverty level affordability ratios increase as system size decreases. Affordability ratios for household in deep poverty follow the same trend. Households earning poverty and deep poverty income levels thus have a higher water bill burden if they are in a very small or small system compared to households with similar incomes in medium, large, and very large systems (Table 3.5).

3.5.4 *HOUSEHOLD POVERTY INDICES AND AR_{CP} & AR_{DP}*

We estimated the percentage of households in systems earning at or below the county poverty (HH_{CP}) or deep poverty levels (HH_{DP}) to indicate the proportion of households within a system facing at least the county poverty or deep poverty level affordability ratios. Nearly all water systems have some percentage of households living at or below the county poverty level (Table 3.6). The median percentage of households at or below the county poverty level in the sample is estimated at 24% (IQR = 15%; 31%) and the median percentage of households at or below the deep poverty is 9% (IQR = 5%; 13%).

The distribution of HH_{CP} and HH_{DP} in the sample of 1,501 systems is not significantly different from that for the overall community water system list (Mann Whitney U-test $p = 0.07$). For very small systems, however, Mann Whitney U test indicates a significant differences in HH_{CP} and HH_{DP} between the sample and full water system list ($p < 0.001$). Mean poverty levels for very small systems are lower in the sample (1,501 systems) compared with the overall system list (Table 3.6)–the means between the sample and full list differ by around 2%. This corroborates finding in the bias assessment (Appendix B6) and discussed above.

System size was significantly associated with household poverty indices for the full system list and for the sample list using Welch's One-Way ANOVA and square-root transformations to ensure normality of residuals ($p < 0.001$). The effect size of system size on poverty levels is very small (eta squared = 0.03; 0.01 for sample list and full system list respectively), indicating that differences in poverty levels across system sizes are statistically significant but somewhat marginal in absolute terms.

Table 3.6 Percentage of households at or below county poverty level and deep poverty level across systems in sample (n=1501) and full community water system list with system boundaries (n=2882), for 2015.‡

	Water systems in Affordability Study (n=1501)		**Full Water System List with Boundaries (n=2882)**	
System Size (People in System)	**Households at or below County poverty level,** HH_{CP} (%)	**Households at or below Deep poverty level,** HH_{DP} (%)	**Households at or below County poverty level,** HH_{CP} (%)	**Households at or below Deep poverty level,** HH_{DP} (%)
Very small (<500)	22 ± 13^{c}	9 ± 7^{b}	24 ± 14^{b}	10 ± 7^{b}
Small (501-3,300)	28 ± 14^{a}	11 ± 8^{a}	28 ± 14^{a}	11 ± 8^{a}
Medium (3,301-10,000)	$25 \pm 11^{a,b}$	10 ± 6^{a}	$25 \pm 11^{a,b}$	10 ± 5^{a}
Large (10,001+)	24 ± 8^{b}	10 ± 4^{a}	24 ± 9^{b}	10 ± 5^{a}

‡ Results are rounded to the nearest integer. For post-hoc tests, all data were square-root transformed to ensure normally distributed residuals, and back-transformed for the table. For each measure shown, means that share the same letter column-wise are not significantly different from one another based on Games-Howell non-parametric post-hoc tests on square-root-transformed data. Post-hoc letters were calculated using *userfriendlyscience* package in R (Version 3.5.1; R Development Core Team) and ordered to start comparisons with the highest mean value.

Small systems serving 501-3,300 people have the highest average poverty levels (mean $HH_{CP} = 28\%$; standard deviation = 14%), but all system sizes have average HH_{CP} greater than 20% (Table 3.6). The household deep poverty index shows relatively similar levels of poverty across system size categories, with very small systems having slightly lower average levels overall. Post-hoc tests for the full water system list indicate significant differences in mean poverty levels between very small and small systems ($p < 0.001$), but no significant difference in HH_{CP} between very small and medium or large systems. As with the sample used in the affordability analysis, the effect sizes are small (eta squared = 0.01). Though statistically significant differences exist across the size categories, the effect sizes of system size on HH_{CP} and HH_{DP} are small (eta squared = 0.03 for both models). Our sample biases very small systems towards slightly lower mean poverty levels(Table B6).

When sensitivity analyses were run on the full system list and the sample list, systems with potentially unreliable income estimates were removed. For the affordability sample less unreliable estimates, very small system mean HH_{CP} levels remain statistically different from mean poverty levels in other system sizes, but differences among HH_{CP} in small, medium, and large systems are no longer significant (Appendix B5). No differences were seen in post-hoc test for HH_{CP} after removing unreliable income estimates from the full water system list with boundaries (Appendix B5). Sensitivity analysis did not affect means or multiple comparison tests for HH_{DP}.

Overall, the household poverty indices demonstrate the context of poverty-level affordability challenges for different systems. Figure 3.2 shows the relationship between poverty-level affordability ratios (AR_{CP}) and the percentage of households in poverty within a system (HH_{CP}). The figure can be used to assess the prevalence of systems at various affordability ratio and household index cutoffs, while keeping in mind the slight bias against very small systems. For example, we see that across system sizes, many water systems have a high percentage of households in poverty and water is relatively affordable (e.g. AR_{CP} less than 1-2%).

We also see that a fifth of all water systems (n = 318) have at least a third of households in poverty (i.e. $HH_{CP} = 33\%$). This proportion is slightly higher in the full list of water systems with boundaries (n = 693 out of 2,882, or 24% of systems, with $HH_{CP} = 33\%$). Of the 318 systems with high poverty levels in the sample, the average county poverty level affordability ratio is 2.1%. This is close to the average AR_{CP} for all systems. While lower than thresholds of 3% advocated for use with disposable incomes like CP, households in poverty paying 2.1% of their income on drinking water alone are likely to exceed the 3% threshold if sewer cost data become available and incorporated into these metrics. This indicates a potentially concerning affordability issue for households in poverty on the one hand, and a substantial fraction of systems (21%) whose customer base is economically vulnerable.

Figure 3.2 also shows that some systems–usually small or very small systems–have high percentages of households at or below the county poverty level (e.g. $HH_{CP} >$ ~10%) and relatively unaffordable water bills (e.g. $AR_{CP} >$ ~3%). Figure 2 illustrates that poverty levels, water bills, and system size vary and produce distinct contexts of water affordability challenges for households in community water systems.

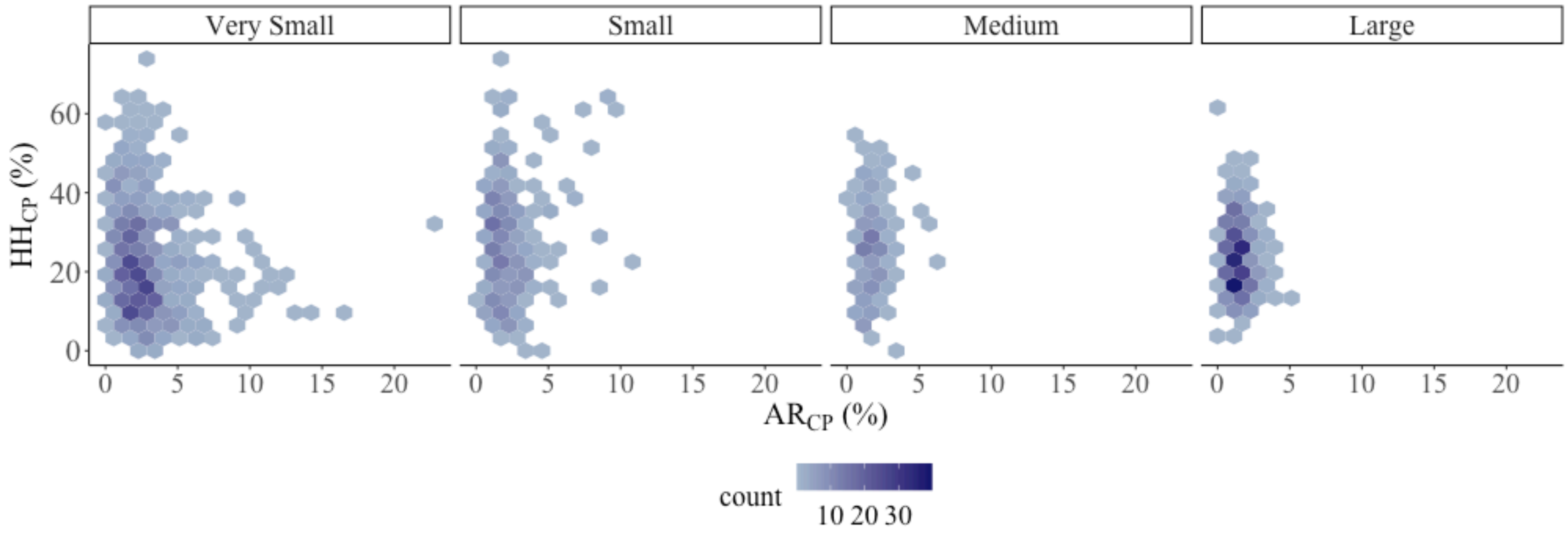

Figure 3.2 Percentage of households below county poverty threshold (HH_{CP}) vs. affordability ratio at county poverty threshold (AR_{CP}), by community water system size (n=1,501).

Scatterplot indicates affordability ratios at the county poverty threshold on the x-axis and households at or below county poverty income levels within the water system on the y-axis. Color fill shows the number of systems in each hexagon. Strip text above plots indicate community water system size. System size is determined by population served: very small = 25-500 people; small = 501-3,300 people; medium = 3,301-10,000 people; large = 10,000+ people.

3.6 DISCUSSION

We developed an approach to measure water affordability in a human right to water context and applied this approach to the case of California's community water systems. This study offers four new contributions to water affordability research. First, we develop affordability ratios by conceptually linking measurement and data choices to broader aims for the human right to water. We consider our measures as relates to spatial scale, basic needs water, economic vulnerability, available income, and criteria for affordability. We demonstrate how these dimensions can be applied as a framework to develop transparent and consistent affordability measures with limited data, and explicitly address the inherent challenges for each measure. Efforts to monitor the human right to water risk perpetuating the illusion of universal access in the U.S. if vulnerable communities are not represented in the measures used. Therefore, our second contribution is the development of two new water affordability ratios for households in poverty, and an improved version of a commonly used ratio for households earning median-level incomes. Thirdly, our analysis is, to our knowledge, the first statewide assessment of water affordability for households served by community water systems in California, and one of the few studies that includes small systems. We innovate on past affordability research by developing multiple ratios, rather than adhering to a single metric or threshold of affordability. This allows us to avoid a binary determination of affordable / unaffordable, when this is–in the end–a political decision for local contexts and groups to determine. Finally, we co-produced this research with a government agency that develops tools to track and monitor California's human right to water. Co-production as a mode of scientific research enhances the relevance of research questions and methods while also increasing the likelihood of research translation into decision-making (Lemos *et al* 2018).

Collectively, we find that water is relatively affordable for households earning at median income levels in a majority of California water systems for which data are available. That is, across these systems, average AR_{MHI} equals 1.1%, which is consistent with cross-national comparisons showing that households in industrialized countries pay on average 1.1% of median income for water (Smets 2009). While AR_{MHI} has sustained critique about its value as a metric, we posit that it is a useful complement to poverty focused metrics. We identified several cases where a water system's median household income was lower than the county poverty threshold–indicating that the AR_{MHI} measure can identify affordability risks in water systems where a majority households are low-income. In low income communities this metric can signal affordability challenges for households *and* the overall community. For example, for the 41 systems in which median household incomes are below their county poverty thresholds, the average affordability ratio is nearly three times the state-wide average (3.1%).

We assess affordability for households consuming 6 HCF per month in an effort to exclude luxury water consumption that is not protected by a human rights framework. While this is defensible by 'basic needs' water requirements and conservation standards, households may in fact need higher water volumes for even basic needs in certain contexts (e.g., households with many members or sick members). Thus water affordability implementation programs may wish to cast a wider safety net for affordability than current measures for monitoring affordability do. For example, the

State Water Boards' proposes using 12 HCF in their proposed LIRA program (State Water Resources Control Board 2019b). Water rate assistance programs might allow for a variable volume of water eligible for rate assistance based on household size to account for larger households or households facing chronic illnesses.

To better represent economically vulnerable groups, we evaluated water affordability for households earning at county poverty as well as deep poverty levels. The median county poverty level affordability ratio was relatively affordable (1.8%), but the impact of water bills is substantial for households earning at the deep poverty level. A quarter of water systems have bills that are 2.7% of county poverty level incomes and 5.3% of deep poverty levels. For economically vulnerable households in these systems, even average water bills ($52.44 per month for 6 HCF) can be unaffordable. These affordability ratios for economically vulnerable households are cause for concern since water rates, especially in urban areas, have been rising faster than inflation (Hanak *et al* 2014) while incomes have stagnated or declined for middle and low-wage earners (Reidenbach 2015, Gold 2015). These trends are projected to exacerbate water affordability challenges across the U.S. (Mack and Wrase 2017).

On average, affordability ratios are highest for households served by very small and small systems at every income level evaluated in this study. This is partially explained by the fact that households served by smaller systems (25-3,300 people) have some of the highest monthly water bills for 6 HCF. Yet even within larger water systems and systems categorized as non-disadvantaged, affordability ratios for households at the county and deep poverty income levels are often close to, or exceed, international and national thresholds for affordability. These findings support previous research in Los Angeles, which identified high concentrations of poverty across system sizes, especially in larger water systems (Pierce and McCann 2015).

Our findings indicate that unaffordable water is not limited to small water systems lacking economic resources. In fact, our approach reveals heterogeneity in affordability challenges for households across and within water systems in the state. Until recently, current state-wide approaches to ensure affordable water have focused on economically disadvantaged systems whose upgrades to comply with water quality standards will push water bills above 1.5% of the water system's median household income (State Water Resources Control Board 2018). Providing financial resources to economically disadvantaged systems is critical, but affordability support should not overlook the reality that households also struggle to pay for water in larger, non-disadvantaged community water systems. Though many of these systems may provide direct subsidies, households are under-enrolled and policies vary across systems (State Water Resources Control Board 2019b). This point is echoed by other work on U.S. affordability challenges that note the prevalence of poverty in large, urban water systems (Mack and Wrase 2017, Teodoro 2018, Pierce and McCann 2015, Colton 2017).

Our study addresses several critiques of affordability ratios, but there remain areas for continued research and improvement. First, a little more than half of the California community water systems (1,501 out of 2,903 community water systems) had adequate data to evaluate water affordability. Most of the missing data came from systems not reporting water bills. More work is needed to fill in the gap for those systems in order to estimate their affordability challenges. Secondly, while we assessed data reliability and applied a sensitivity analysis for data that we flagged as unreliable due to high sampling error in the American Community Survey, we did so only for water systems falling within one block group (504 systems of the 1,501 in the study). For the remaining systems, we aggregated block group level data to the water system service area, which theoretically should increase data reliability by increasing the effective sample size of estimates. More research is needed to assess data reliability of census estimates in new geographies that do not overlap with census boundaries.

Thirdly, there is an under-representation of very small systems in the final study list (Appendix B6). This is largely due to missing water bill data. We included measured confounders of missingness in a generalized model of affordability to investigate water bills by system size while minimizing bias. However, future work on water bill data in California should consider new randomized sampling efforts to collect a representative sample of smaller systems where the full dataset is not available. County poverty levels for multiple county groups are a limitation of data scale that likely biases against better data resolution for smaller, rural water systems. Together, these limitations indicate broader trends in research where smaller systems with poor data are underrepresented (Balazs *et al* 2012). We hypothesize that our sample underestimates the affordability challenge in California because there are fewer lower-income systems represented. Finally, the income constraints across affordability ratios are estimates that should be interpreted with care. Median income levels do not account for taxes or other expenditures, and so underestimate the extent of the affordability problem for median-income households. California poverty thresholds could underestimate disposable incomes for households because they do not include benefits such as housing subsidies, but may also overestimate real disposable income because expenses for non-water essentials are not removed.

More research is needed to assess dimensions of water affordability that are not captured in ratio-based approaches. Examples include assessing connection fees (Kessides *et al* 2009) or well maintenance costs, and tracking water shut-off consequent to the lack of means (OHCR 2014, United Nations Committee on Economic Social and Cultural Rights 2002). At the same time, the affordability challenges facing particularly vulnerable communities/people also needs attention, such as people living without houses, mobile home park residents in vulnerable systems, households served by systems with fewer than 15 connections ("state smalls"), and private well-owners. These are all populations that do not have representation in current affordability ratio approaches (Pierce and Gonzalez 2017b, Environmental Law Clinic 2017, Moore *et al* 2011) or in this study. Furthermore, the impact to households forced to buy bottled water due to poor tap water quality are not factored in to our proposed affordability ratios. Previous work has also shown that increased water costs, the risk of poor water quality, low water-system financial capacity, and high concentrations of low-income communities of color–particularly in unincorporated communities–are entwined (London *et al* 2018, Balazs and Ray 2014). Community mistrust in tap water has also been identified as a driver of bottled water consumption, with the resultant time and money costs falling disproportionately on communities of color (Javidi and Pierce 2018). These multidimensional aspects of affordability demand better data, additional metrics, and increased representation marginalized groups in human right to water monitoring efforts.

Future research goals for this work therefore include investigating the extent to which household water affordability relates to ethnic or racial disparities and to human right to water pillars such water quality and accessibility. Such work will sharpen our understanding of affordability and also inform policy to progressively and equitably realize the human right to water in California.

CHAPTER 4

4. CLIMATE IMPACTS OF BOTTLED OVER TAP WATER IN EMERGING ECONOMIES–INSIGHT FROM MEXICO[3]

Abstract

Bottled water rivals municipally treated water as a drinking water source in many countries around the world, yet its climate impacts are not well studied outside of Europe and the U.S.. Mexico has the highest per capita consumption of bottled water globally, and 20 liter plastic bottles (known in Mexico as garrafones) are the primary drinking water source for most households. Here, we model embedded greenhouse gas emissions over the lifecycle of an average garrafón by creating an input-output model for the energy requirements of bottled water over seven stages: 1) plastic resin production; 2) bottle fabrication; 3) plastic cap fabrication; 4) water source (municipal, well, or trucked water); 5) water treatment in a treatment plant; 6) bottle cleaning and filling; and 7) bottled water distribution. Mexico's garrafón market reflects household bottled water use patterns in low and middle income countries globally, where bottles are large in volume, frequently reused compared to disposable bottles, and product distribution patterns are diverse. Moreover, the market share of micro- to small-scale treatment and bottling plants is high. We find lower emissions per cubic meter of garrafón water relative to disposable plastic bottles, with most emissions coming from distribution of bottles as opposed to plastic resin due to high reuse. We estimate a baseline garrafón to have 46 times the global warming potential of municipal tap water treated by a household water treatment system and 168 times more emissions intensive than municipal tap water. Were households to switch from garrafones to household treated tap water, we estimate savings nationwide could be as large as 1-4.7 million tonnes of CO_{2eq} per year.

[3] This chapter will be submitted to a journal in edited form and has been approved for use in this dissertation by my co-authors, Fermín Reygadas, Diego Ponce de Leon Barido, and John Pujol.

4.1 INTRODUCTION

Households with and without tap water access are consuming water sold in plastic bottles at an increasing rate, especially in low-middle income countries (Cohen and Ray 2018). Global bottled water consumption sales have grown at a rate of 6.2% per year since 2013 (Rodwan 2019). Distrust in tap water (Pierce and Gonzalez 2017a, Javidi and Pierce 2018), intermittent supply, and organoleptic (taste, smell) factors (Doria *et al* 2009) are documented drivers of bottled water use among those with and without improved water access globally. While an estimated 71% of the world's population now has access to safely managed drinking water–up 11% since 2000– there are still 785 million people without water access (United Nations 2019b). Sustainable Development Goal (SDG) 6 set an international agenda to close this gap, but metrics to benchmark and track the sustainability of water access in households are underdeveloped. This is especially relevant since 2016, when the Joint Monitoring Programme recognized "packaged" water as a growing primary drinking water source. Since this point, SDG 6 has included access to bottled water as an improved source (World Health Organization 2017).

In this chapter, we investigate climate impacts of bottled water by developing a metric for the global warming potential of bottled water use in urban Mexico. Mexico leads the world in per capita consumption of bottled water (Rodwan 2019). While global bottled water consumption is dominated by single-use plastic bottles, reusable 20 liter home and office delivery bottles, known as garrafones, make up two-thirds of consumption in Mexico (Rodwan 2017). An estimated 76% of urban households use garrafones or bottled water of other sizes as their primary drinking water source at home (INEGI 2017). Earlier work estimated that 80% of households in urban areas used garrafones, with smaller percentages drinking filtered tap (8%) or disinfected (2%) water at home (IADB 2010). Purchasing bottled water is more expensive than tap water, posing equity concerns.

To our knowledge, the garrafón has undergone no analysis or study for its environmental impact in Mexico, or any other large emerging economies. The availability of data and diversity of use cases for bottled water in Mexico make it a compelling case study for understanding the climate impact of bottled water as its growth increases around the world. The carbon footprint of smaller disposable water bottle use has been analyzed in various life cycle assessments (LCA) in North America and Europe (Quantis 2010, Jungbluth 2005, Shen *et al* 2010, Franklin Associates 2009b, Nessi *et al* 2012). These studies provide relevant insight and data for bottled water manufacturing and materials processes, but they lack specificity for low and middle-income countries, where garrafón re-use rates are high and bottling manufacturers are abundant.

We develop a life cycle assessment model to estimate a country-wide average estimate of garrafones' global warming potential (as kg CO_2eq/m^3 per garrafón) across Mexico. We develop several scenarios to test the sensitivity of our results. Data on the distribution of garrafones between bottling plants and households is limited, but essential to model-building efforts. As such, we elicited expert input from local water bottlers and distributors to inform model assumptions. We compare our results to estimates of life cycle emissions of tap water and household water treatment from previous research (Reygadas *et al* 2014) to evaluate the emissions trade-offs among these three household water use scenarios.

4.2 METHODS AND DATA

Life cycle assessments quantify direct and embedded energy and various environmental impacts of a product from its production to end-of-life, or "cradle-to-grave". Applying principles of mass and energy balance to a process or product, LCA enables benchmarking for product sustainability and the identification of key inputs or processes that may be environmentally harmful (Garfí et al 2016). LCAs require definition of system scope, an inventory of processes, an assessment of select environmental impacts, and interpretation of results. The present study is assessment of garrafones, from production to use within the household; data on recycling and waste production within Mexico was substantially limited and therefore this study reflects a "cradle-to-use" rather than "cradle-to-grave" estimate of life cycle emissions. We focus specifically on global warming potential as carbon dioxide emission equivalents to represent climate-related impacts.

Previous LCAs on bottled water found global warming potential impacts of 127 kg CO_2eq/m^3 to 425 kg CO_2eq/m^3, with home and office delivery bottles (i.e. garrafones) representing the lowest impact due to high bottle reuse (Table 4.1).

Table 4.1 Literature review of previous life cycle assessments on plastic bottled water.

Presented here are baseline scenarios modeled within each study; assumptions differ widely by study; this table highlights some of the main assumptions.

Source	Location, Year	Model scenario[1]	Functional Unit[1] (L)	CO_2 equivalence[1] (kg/m^3)	Energy equivalence[1] (MJ/L)
Jungbluth	Switzerland, 2005	Cradle-to-gate; non-reusable Cradle-to-grave; non-reusable	1 1	178-425[2] 161.6	4.23-8.34 ~4.18
Franklin & Associates	Oregon (US), 2009	Cradle-to-grave; 40 uses	18.9	126.7	~1.96
Gleick & Cooley	California (US), 2009	Cradle-to-gate; no recycling	1	Not available	5.6-10.2
Quantis for Nestlé	North America, 2010	Cradle-to-grave; non-reusable Cradle-to-grave; 100 uses	0.5 3	274 350	Not available
BIER Europe BIER US	Europe and US, 2012	Cradle-to-grave; 3% recycling credit	1.5 0.5	108.5 165.6	Not available
Nessi et. al	Italy, 2012	Refillable 1L Polyethylene terephthalate bottle	152.1	163	Not available

[1]Literature unit values were converted to Liters water, kg CO_{2eq}, or MJ/L for comparison with the present study.
[2]High end value reflects an assumption of long transportation distance in distribution.

Several aspects of garrafón use are unique to Mexico. Among these are: treatment plant treatment energy requirements, market share data, distribution pathways, bottle reuse rates, and recycling options. These steps are discussed below; but assumptions vary widely across the literature and suggest large influences on metric outcomes.

To capture the large range of potential scenarios that impact the lifetime emissions of garrafones, we developed three emissions scenarios: low, baseline, and high. In the sections that follow, we present the life cycle analysis scope, boundaries, and functional unit followed by an extensive discussion of assumptions for scenario building in the model. Ultimately our process based LCA integrates an extensive review of literature assumptions, augmented by twenty informal interviews with industry professionals in Mexico to bound our assumptions.

4.2.1 *Life Cycle Analysis Scope, Boundaries, and Functional Unit*

The scope and boundaries of our life cycle analysis included the following stages of production and distribution:

- Producing plastic resin,
- Fabricating the reusable bottle,
- Fabricating the single-use plastic closures,
- Extracting & transporting water (municipal, well, or trucked water),
- Treating the water in a treatment plant,
- Cleaning and filling the bottle,
- Distribution of garrafones, and
- Energy intensity of transportation fuels and electricity

Error! Not a valid bookmark self-reference. shows each stage and the study boundary. A key distinction to note is that while transportation of materials occurs in every stage of production, the distribution stage refers to the transport of full garrafones from plant to home (potentially via a stop for storage at a store or distribution center). The distribution of garrafones in Mexico is distinct from previous studies (due to a high volume of treatment plants) and thus emissions from transportation during the distribution phase are modeled and discussed in detail below. Importantly, the transportation associated with the return of empty garrafones is embedded in the distribution phase, because transit distances garrafón distribution are round-trip. When bottlers deliver full bottles, they also pick up empty ones to bring back for washing and filling.

The garrafón is typically sold in 20-liter containers. For data manipulation, the main units used throughout the analysis are one tonne for material inputs and one cubic meter (m^3) for water inputs. The functional unit for the system is one cubic meter.

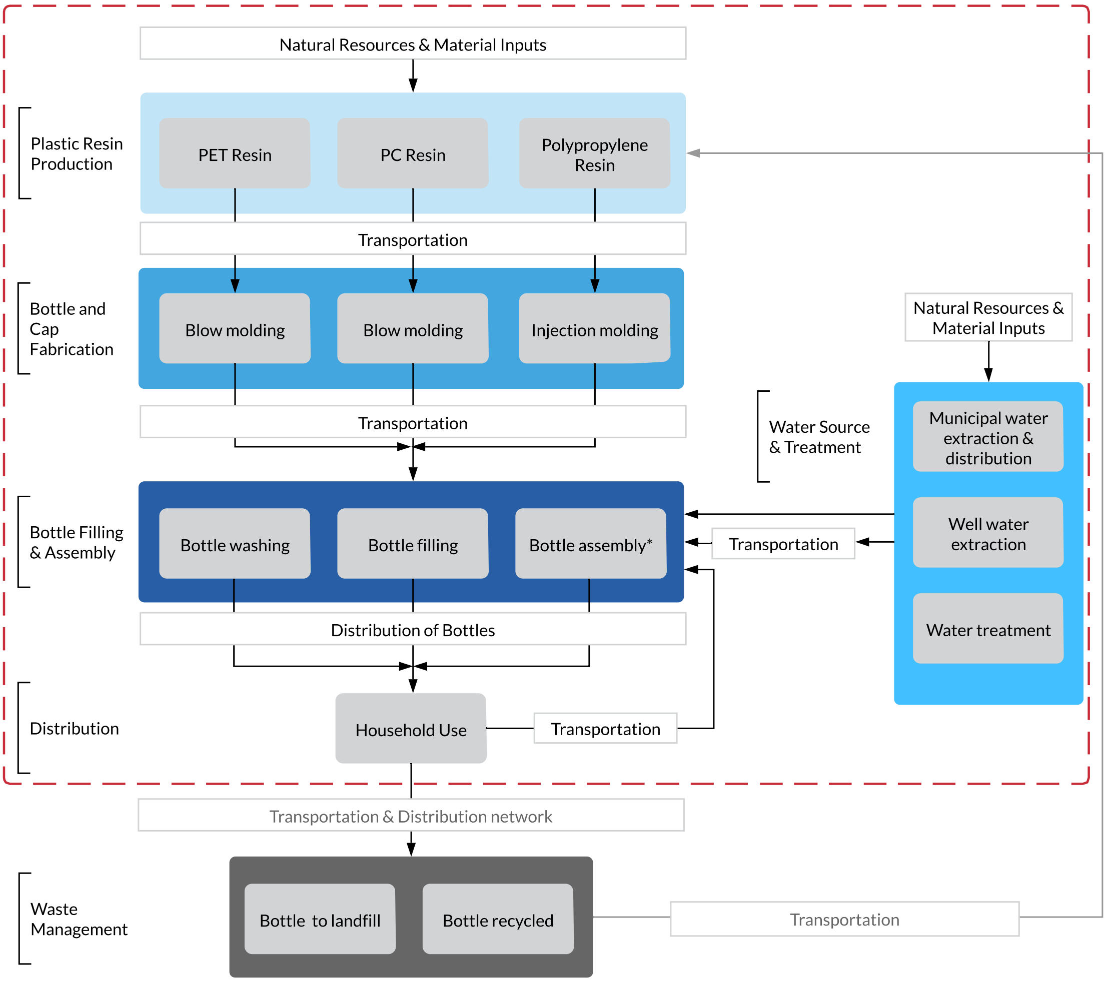

Figure 4.1. Schematic of life cycle processes for bottled water and system scope.

Red-dotted line denotes system boundaries. PET = Polyethylene terephthalate; PC = Polycarbonate; PP = Polypropylene. *Bottle assembly includes adding labels, which are not included in the assessment.

The model did not include emissions associated with several processes due to an assumed minimal emissions contribution over the life cycle of a garrafón or a lack of relevance in the Mexican case:

- Emissions associated with labeling and stickers are found to be minimal in the literature and thus are not included in this study. Gleick and Cooley (2009) report a labeling and sealing of 0.01% to 0.18% of the total bottled water energy intensity (Gleick and Cooley 2009).
- The embedded emissions of infrastructure or machinery in all stages of the garrafón production are not included. Capital equipment emissions are distributed among thousands and often hundreds of thousands of cubic meters of water, thus the emissions are anticipated to be minimal per garrafón.
- Space conditioning in buildings like supermarkets, convenience stores, or treatment plants are only likely to influence some portion of the garrafón market

(for example, those sold in convenience stores), however these spaces house and sell many more products than the garrafón and thus an allocation of emissions to the garrafón is anticipated to be minimal.

- We do not consider cooling and chilling emissions in this life cycle assessment. Chilling or cooling of the garrafón is uncommon in Mexico, as they are mostly sold at room temperature.
- We do not include home-washing because bottle washing occurs at the bottling plants where the garrafones are refilled.
- Unlike bottled water in smaller bottles, garrafones do not require additional disposable cardboard and plastic packaging during transportation.
- Vessels used to drink water out of the garrafón will also be used to drink water from the tap or treated at the household, therefore it is not a useful addition for a comparison among garrafones and other sources of drinking water.
- Activities of employees for water bottling plants, such as commuting, are not included in the scope of this analysis.

Finally, we did not incorporate recycling credits or emissions associated with recycling or landfill. After a bottle's lifetime (perhaps 100 uses), it may be recycled or sent to landfill. Emissions from recycling can enter an LCA if the study bounds include the process of recycling itself (end of life) or if the study bounds include the use of recycled plastic in the virgin product (beginning of life). Studies inclusive of recycling distinguish the recycled product from the virgin product as a "second life" process wherein the post-consumer waste is not counted against the recycled product, or the virgin product is allotted recycling credits (Frischknecht *et al* 2007, Shen *et al* 2010).

Sources and data on recycling rates for both beginning-of-life and end-of-life of garrafones are highly variable. For the former, interviews with U.S. bottlers, conducted in the Franklin Associate study found that home and office delivery bottles (approximately the same bottles as garrafones) are not using recycled material (Franklin Associates 2009b). Given high levels of import of plastic resin from US to Mexico, it is likely that recycling credits associated with the plastic resin phase would be minimal. For recycling associated with end-of-life, we would require data on recycling rates and associated energy costs. One report estimates a countrywide recycling rate of around 17%, with 56% of polyethylene terephthalate (PET) being recycled and 60% of that plastic staying within Mexico (ECOCE 2019), but little of this is likely recycled into new garrafón production. Previous work indicates minimal relative emissions from the recycling process in smaller plastic bottles (Gleick and Cooley 2009). We did not estimate recycling or landfill end-of-life emissions due to lack of adequate data for garrafones in Mexico.

4.2.2 *Methods to Estimate Life Cycle Emissions of Garrafones*

We adapted energy requirement estimates of garrafón production processes from a study of home-office-delivery bottles in Oregon, USA, by Franklin Associates (2009). The Franklin Associates study was the most comprehensive dataset available, and allowed us to use energy, transportation, and electricity values associated with all material inputs and embedded energy for: PET and PC resin, polypropylene (PP) caps, fabrication processes (blow molding and injection), drinking water treatment, and bottle filling and washing (Franklin Associates 2009a). In every step of materials production there are embedded material inputs and transportation (e.g. transporting PET resin pellets from a resin plant to a bottle fabrication facility).

Many characteristics of the garrafón market are not well documented; for example, little to no published data exists for bottling plant sizes, production volumes, or distribution routes. As such we consulted with several plastic injection experts, water treatment plant operators, refill stations, and distributors across Mexico. Interviewees provided knowledge about garrafón water sources, consumption of electricity by bottling plants, distribution distances, and the frequency of bottle reuse, among other parameters. Our interviews were informational and selected based on convenience sampling of contacts our team had in the states of Mexico, Baja California Sur, and Chiapas. Our aim was not to develop a comprehensive set of assumptions but rather to help us bound assumptions about: market share of plastic types, vehicle efficiencies, distribution patterns, water sources, and water treatment in Mexico.

Of the twenty experts we spoke to, we formally collected data associated with bottle filling, treatment, and distribution of product from nine operators (four bottling plants and one refill station in Chiapas; four bottling plants in Baja California Sur). We spoke to treatment plant operators whose main energy inputs would vary substantially in terms of salinity and total dissolved solids (TDS). While there is likely high variation in energy requirements for treatment plants across the country, high salinity of intake water is known to have a large impact on the energy intensity of treatment operations (Cooley and Wilkinson 2012). This is due in part because reverse osmosis is the most energy intensive treatment step, and plant operators must replace membranes more frequently when water is highly saline (Stokes and Horvath 2006). Baja California Sur has lower water availability and higher TDS than Chiapas (Reygadas 2014), providing us with a bound of very low and very high energy costs associated with water source inputs and treatment processes. Distribution patterns and production volumes (from 20 to 2,300 garrafones per day) also differed between states, providing us with a better view of market variability.

The final model was built in Microsoft Excel and draws from a life cycle garrafón production input-output library (a life cycle inventory). The input-output library converts estimated material use (e.g. tonnes of plastic resin) into greenhouse gas emissions (i.e. carbon dioxide equivalents) for each step of the garrafón production process using conversion factors. Through this model we explore various scenarios to test assumptions about water source, weight of bottle, plastic type, distribution distance, vehicle types, and bottle reuse values, among other parameters.

We report a baseline metric (kg CO_2eq/m^3) using weighted average emissions factors for key inputs. To capture a range of possible emissions outcomes, we used lower and upper quartile values of key inputs for which we have a range of data (i.e. conversion factors for water source emissions and electricity). Where this was not available, assumptions about low/high scenarios were made based on interviews, literature, or– in the case of distribution scenarios–values intended to test the sensitivity of the model with low and high bounds.

4.2.3 *GLOBAL CONVERSION FACTORS*

4.2.3.1 GLOBAL WARMING POTENTIAL

Life cycle assessments require several conversion factors to translate material inputs into homogenous units to estimate global warming potential in kilograms of CO_{2eq}. We use international-standard data on global warming potential factors to normalize emissions estimates across multiple greenhouse gases. Specifically, we estimate cumulative CO_2 equivalence (kg CO_2eq) for all life cycle phases from carbon dioxide (CO_2), methane (CH_4), and nitrous oxide (N_2O) (Parry *et al* 2007) using the 100-year global warming potential factors (US EPA 2011), where N_2O and CH_4 have global warming potentials 265 and 25 times that of CO_2, respectively.

4.2.3.2 FOSSIL FUEL CONVERSION FACTORS

Both direct and indirect emissions are important in the life cycle of bottled water. Indirect emissions are those associated with the extraction, transportation, refining, product distribution, and storage of fossil fuels inputs, whereas direct emissions are associated with fuel combustion at any stage of the life cycle. While a conversion inventory exists for multiple fuels in the United States and Mexico in terms of kg CO_2eq/unit fuel, it only contains direct emissions of combustion (US EPA 2011, Pardo *et al* 2008). We therefore used the United Kingdom Department of Energy and Climate Change (DECC) and the Department for Environment, Food and Rural Affairs (DEFRA) data on direct and indirect emissions conversion factors for fuels. DECC and DEFRA data is compiled under the GHG Protocol for industry and company emissions reporting (Hill *et al* 2013) in an emissions calculator that provides detailed fossil fuel emissions factors estimates.[4] See Appendix C for discussion of indirect and direct emissions and a literature review of emissions factors (Table C1).

4.2.3.3 ELECTRICITY EMISSIONS FACTOR

We estimated a nationwide electricity conversion factor of 0.57 (IQR = 0.29, 0.72) CO_2eq/kWh for Mexico based on previous work (Reygadas *et al* 2014), which takes the energy composition of the Mexican grid (Santoyo-Castelazo *et al* 2011) and estimates an average metric based on state-level emissions, weighted by installed capacity of each state.

4.2.4 *MODEL SCENARIOS AND DATA OVERVIEW*

Below, we review assumptions and data inputs based on the Mexican case and refer readers to the model and Franklin Associates (2009) for specific energy requirements for plastic resin and plastic cap fabrication (Franklin Associates 2009b). After introducing these assumptions, we summarize the key input factors in the model for

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4 Access to the calculator and its appendixes can be found here: https://www.gov.uk/government/publications/2012-greenhouse-gas-conversion-factors-for-company-reporting. While the instructions warn against using the calculator overseas, the only data taken from the model was for fossil fuel conversion factors by weight and volume, which should not vary substantially by country.

each stage, as well as our assumptions for baseline, low, and high scenarios based on existing literature and interview data.

4.2.4.1 Plastic Resin Production: Bottle Material

Garrafones are made of polyethylene terephthalate, polycarbonate, polyvinyl chloride, or glass. We limit our study to PET and PC bottles as they are the dominant material source for garrafones and few bottlers we interviewed used glass or polyvinyl chloride material.

4.2.4.2 Market Share of Treatment Plants

We referenced available data on market share for bottled water companies based on national statistics and consulting reports that there are nearly 20,000 small to micro-purifying plants (INEGI 2014) and that these comprise nearly 52% of the market-share, on average (EuroMonitor International 2016). Previous sources claim that 85% of bottled water produced by companies considered to be small or micro in scale (Carranza and Vega 2011). The prevalence of small/micro size plants compared with larger distributors is likely to vary widely by region–with one survey in 2011 finding households purchase from non-brand, small/micro plants, from 3% to 70% across districts in Mexico City (Montero-Contreras 2016).

We estimated emissions for two production plant scales, small/micro and large. Given the wide range of potential market shares and the lack of consistent data, we assume small/micro plants comprise 52% of market share for the baseline case, and 80% and 40% of the market share, for low and high emissions scenarios, respectively. We use the market share data to weight emissions from these two types of plants in our composite metric. For example, larger companies use combination trucks to distribute garrafones across larger distances and thus the emissions associated with these activities are weighted relative to estimated market share of large companies with combination truck distribution.

Large Scale

We referenced Nestle, BIER, Franklin & Associates, and Gleick and Cooley to establish baseline scenarios for the energy intensity of water treatment and distribution routes (of filled bottles) for large-scale treatment plant operations (Quantis 2010, Beverage Industry Environmental Roundtable 2012, Franklin Associates 2009b, Gleick and Cooley 2009).

Small/Micro Scale

To shape assumptions for micro/small size garrafón plant operations, we relied on 20 informational interviews with water treatment plant owners in the states of Chiapas and Baja California Sur. Nine plants provided data on water treatment energy requirements, distribution distances, garrafón deliveries, and vehicle efficiencies. The aim of these interviews was to elicit upper and lower bounds of energy intensities in micro-to-small scale plants to inform the potential range of scenarios across different states in Mexico.

4.2.4.3 Emissions of Water Source

We use emissions conversion factors for municipal water with data from the Mexican grid based on previous work, where we measured the energy intensity per cubic meter of water of 0.46 (IQR = 0.29, 0.67) kWh/m^3 based on a weighted average of state-level estimates for electricity capacity and annual water consumption (Reygadas *et al* 2014). In combination with the Mexico-specific electricity emissions factor presented above, this yields a municipal piped water emissions factor of 0.26 (IQR = 0.07, 0.44) kg CO_2eq/m^3.

Well pumping emissions were calculated based on electricity inputs from Franklin Associates (2009) and compared with interview data. Treatment companies also source water from private wells and water delivery trucks (known as pipas). Interviews with nine bottling companies provided data for fuel consumption for water delivery by trucks, which were then added to the emissions estimate for pumping well water into the trucks (Table 4.2).

Table 4.2 Direct emissions associated with water source.
Estimated emissions factor for water source modeled for baseline scenario.

Water Source	Direct Emissions (kg CO_2/m^3)	Source
Trucked	2.07	Interviews
Municipal	0.26	Reygadas et al. 2014
Well	0.31	Franklin Associates 2009a

4.2.4.4 Emissions of Water Treatment Processes

The range of treatment operations in Mexico's bottled water industry matches those used in most regions of the world, where bottled water plants provide additional treatment to municipal, well, or trucked water (filtration, ultraviolet, ozone, and/or reverse osmosis) (Gleick and Cooley 2009, Centro Mario Molina 2011).

Filtration consists of the basic processes of primary water treatment, including pumps to convey water through screens, filters, and coagulation basins. At the municipal plant scale, screens and filtration devices are usually placed after source water has been coagulated and settled. We used electricity requirement estimates from (Franklin Associates 2009a) for all treatment processes except filtration and reverse osmosis– which we estimated based several sources Table 4.3.

We included a water efficiency factor in our model to account for the production of brine in reverse osmosis processes. Based on (Cooley and Wilkinson 2012) and interviews with small treatment plants, we selected a baseline efficiency factor of 75%, with low and high emissions scenarios of 85% and 40% efficiency, respectively. Direct emissions from each water treatment process are shown in Table 4.3, using the weighted electricity emissions factor for Mexico of 0.57 kg CO_2eq/kWh.

Table 4.3 Direct emissions associated with water treatment.
Estimated emissions factors for baseline model emissions intensity of Mexican energy grid.

Treatment Process	Direct Emissions ($kg\ CO_2eq/m^3$)	Source
Filtration & Plant Electricity	0.43	Reygadas et. al (2014)
RO	0.61	Gleick & Cooley (2009); Cooley & Wilkinson (2012)
UV	0.01	Franklin Associates (2009)
Ozone	0.10	Franklin Associates (2009)

4.2.4.5 Bottle Reuse

After being used in a household, garrafones are returned to bottling plants, where they are washed and re-filled. The transportation emissions associated with this return-trip are accounted for in the distribution of filled garrafones, for which transit distances reflect a round-trip (discussed below). Emissions from plastic resin production are divided by the number of times the bottle is re-used to reflect the effective emissions of plastic resin production over the life cycle of the garrafón.

From interviews with bottling companies, we found that garrafones are reused between 30 and 300 times. Based on these data and industry estimates, we assume a baseline of 100 reuses for PET and PC bottles before they are discarded. Bottle filling and washing for returned garrafones contribute a minor proportion of emissions to the overall process, but are included in the model based on Franklin Associates (2009) data.

4.2.4.6 Emissions of Materials Distribution and Garrafones Distribution

As with any product, there are emissions associated with transportation to produce the product (materials distribution) and emissions associated with the transportation of the final product to a household (garrafones distribution). For materials distribution, we model emissions associated with transportation of: resin materials to bottling companies, plastic bottles to bottle treatment plants, and embedded transportation emissions for the raw materials that go into each of these phases. For the latter, we use data from Franklin Associates (2009) to capture embedded emissions of raw materials at every phase of bottle fabrication. Below we outline assumptions for the other two routes.

Less data is available for transportation requirements in the distribution of garrafones between treatment plants and the home. Literature values on transportation between treatment plants and households range from 80-3,000 km to account for local and international distribution markets (Quantis 2010, Franklin Associates 2009b, Gleick and Cooley 2009, Beverage Industry Environmental Roundtable 2012). Mexico's transportation distances are likely to be much shorter than scenarios modeled in the literature given the prevalence of small/micro treatment plants. Below we outline assumptions around the transportation associated with transport for resin to bottling companies, bottling companies to treatment plants, and the distribution route for garrafones from bottling plant to homes.

Materials distribution: Resin production to bottling companies

After plastic resin is produced, it is transported to a bottling company. Mexico is one of the largest importers of PET and PC bottles from the United States, but plastic resins are also produced in Mexico (International Trade Administration 2019). We could not identify specific data for the proportion of PET or PC bottles using resin produced in the US versus Mexico, but interviewees and export data suggest that a substantial amount of PET resin is produced in Mexico (International Trade Administration 2019), but that plastics for larger bottles (especially PC resin) are imported from the US (Crain Communications 2017).

We therefore use an approximate travel distance of 3,000 km in conjunction with an emissions per tonne-km value (0.053 kg CO_2eq/tonne-km) to simulate a baseline travel distance between the U.S. and Mexico (National Energy Technology Laboratory (NETL) 2008, AEA for DECC and Defra 2012, Schipper *et al* 2010). This distance could also approximate distances within Mexico, given that there are only ten large rigid plastics producers for PET in Mexico. We model a low emissions scenario (1000 km) to approximate a shorter national resin production distance, and a high emissions scenario (5000 km) to approximate a long-distance U.S.-to-Mexico resin production to bottling company scenario. Total emissions from this phase are divided by the number of times a bottle is reused.

Materials distribution: Bottling companies to treatment plants

We use a baseline estimate of 100 km to estimate emissions of transportation between bottle fabrication and treatment plants in Mexico based on interviews and field-knowledge. As garrafones occupy a relatively large volume, their transportation is expensive and most treatment plants source their containers from nearby bottle manufacturers. Total emissions from this phase are divided by the number of times a bottle is reused.

Garrafón Distribution: Treatment plants to home via one or two stops

Once filled and sealed, garrafones are distributed by a variety of routes to homes. Figure 4.2 Distribution pathways of garrafones. shows the combination of distribution routes considered to be representative of most garrafón distribution and vehicle options based on interviews with truck drivers and bottling plant owners. Distances traveled, types of vehicles used, and the number of garrafón deliveries with each trip are based primarily on interview data and anecdotal knowledge from the field (e.g. that a household would pick up two garrafones when traveling to a local treatment plant). We then applied market share data of bottling company size to estimate the likely ratio of each possible distribution route and vary these assumptions to explore low, baseline, and high emissions scenarios.

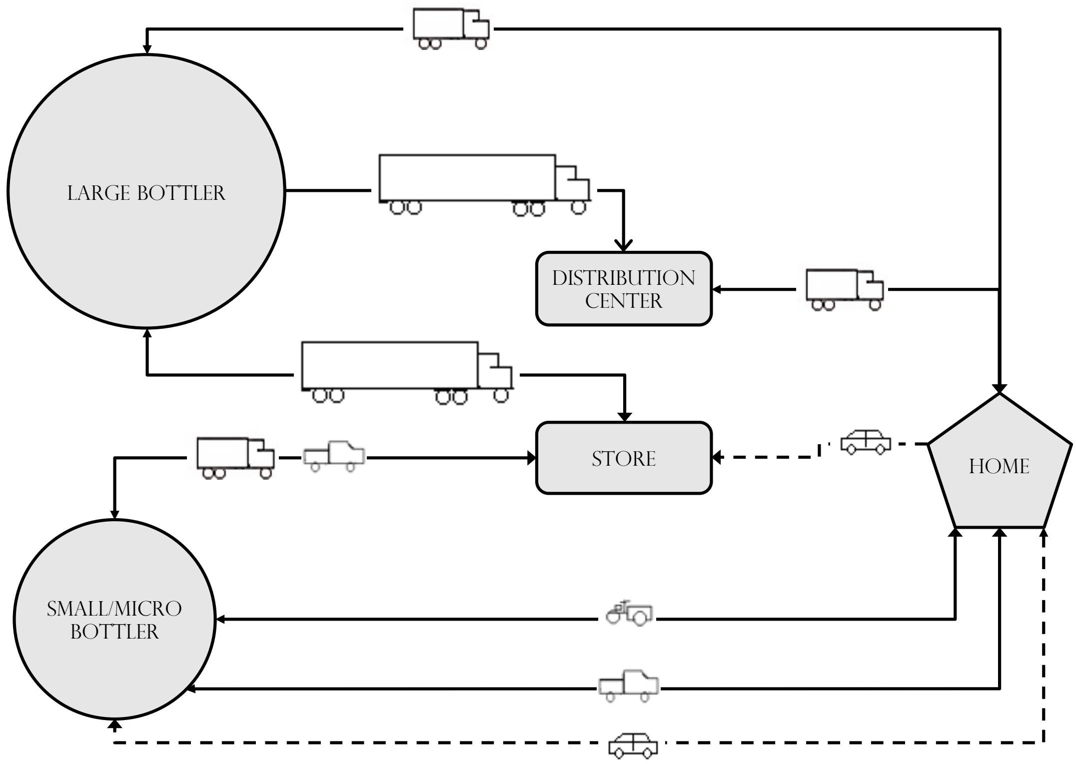

Figure 4.2 Distribution pathways of garrafones.
This scenario map shows the pathways modeled for filled garrafones from large and small/micro treatment and bottling plants to the home. Solid lines indicate a route initiated from the plant; dashed lines indicate a route initiated from the home. All routes model a round trip.

Vehicle fuel efficiency and size determine the ratio of fuel and emissions from distribution that are attributed to each garrafón. We assume that the embedded emissions associated with the production, maintenance, and disposal of vehicles contribute to 10% of the total emissions of the distribution phase (Hill *et al* 2012). We list several prominent trucks used commonly in Mexico with their fuel efficiencies and associated emissions (AEA for DECC and Defra 2012) in Table 4.4. Tricycles are a common distribution route in urban areas, and we model these as having no associated emissions. All distances are estimates of roundtrips in this phase, and implicitly estimate the emissions associated with the delivery of a filled bottle and return of an empty bottle for refill.

Table 4.4 Fuel carrying capacities and fuel efficiencies used in distribution model for garrafón life cycle.

Vehicle	Vehicle Efficiency (km/l)	Emissions (kg CO_2eq/m^3)
Petrol Pickup	10	2.78
Diesel Truck	3	3.24
Diesel Combo	1.5	3.24
Petrol Car	12	2.78
Tricycle	N/A	0.00

Emissions associated with in-store purchase of garrafones are attributed differently than those distributed by trucks delivering only garrafones. When a consumer purchases a garrafón at a store, the emissions from their trip should not be fully allocated to the garrafón purchase. We assume that a consumer is either purchasing a garrafón and other groceries (50% allocation); purchasing groceries and decides to pick up a garrafón (5%); or purchasing a garrafón and decides to purchase other items (95%). Consumers sometimes purchase garrafones directly from the bottling plant, in which case 100% of the trip is allocated to the purchase of bottled water.

4.2.5 *SUMMARY OF ASSUMPTIONS*

Table 4.5 summarizes the key model inputs to produce baseline, low, and high emissions estimates for a countrywide metric of kg CO_2eq/m^3 garrafón water.

Table 4.5 Garrafón model key inputs and scenario parameters.

Factors	Input	Baseline scenario (low emissions – high emissions scenarios)	Unit
Electricity	Electricity Emissions Factor	0.57 (0.29-0.72)	kWh/m^3
Bottle Material	Bottle Weight	0.00075 (0.0006-0.0009)	tonnes
Market Share of Bottle Material	PET Bottles[1]	75 (60-90)	%
	PC Bottles[1]	25 (45-10)	%
Market Share of Treatment Plants	Large Bottler	48 (20-60)	%
	Micro/Small Bottler	52 (80-40)	%
Water Source	Water Source: % Trucked	25 (15-45)	%
	Water Source: % Well	25	%
	Water Source: % Municipal	50 (60-30)	%
Water Treatment	Municipal Water Emissions	0.26 (0.13-0.44)	kg CO_2eq/m^3
	Water Efficiency	75 (85-40)	%
	Reverse Osmosis	50 (20-80)	%
	Ozone	60 (30-80)	%
	Ultraviolet	90 (70-100)	%
	Filtration & Electricity	100	%
Bottle Reuse	Reuse PET[1]	100 (300-40)	#
	Reuse PC[1]	100 (300-40)	#
Customer Allocation	Customer Allocation	50 (5-95)	%
Materials Distribution	Plastic to Bottle Fabrication	3000 (1000-5000)	km
	Bottle Fabrication to Treatment Plant	100 (30-400)	km
Product Distribution (Large Scale)	Plant to Distributer (DC[2]) to Home (DT[2])	10 (5-20)	%
	Plant to Store (DC) to Home (Car)	45 (60-50)	%
	Plant to Home (DT)	45 (35-30)	%
Product Distribution (Small Scale)	Plant to Store (PT[2]) to Home (Car)	25 (40-20)	%
	Plant to Home (PT)	45 (30-40)	%
	Plant to Home (Car)	20 (15-35)	%
	Plant to Home (T[2])	10 (15-5)	%

[1]PET = Polyethylene terephthalate; PC = Polycarbonate Polyethene
[2]Abbreviations reflect different transportation modes. DC = Diesel Combination Truck; DT = Diesel Truck; PP = Petrol Pickup Truck; T = Delivery Tricycle.

4.2.6 *Nine LCAs based on Field Data*

Interviewees enabled us to bound upper and lower assumptions for bottling operations, water sources and treatment, as well as distribution routes. This helped shape assumptions where data was very scarce. As discussed above, these included assumptions about: the type of plastic used to create bottles, bottle reuse, trucked water emissions, the types of treatment technologies used (e.g. the share of ozone versus UV), and the common distribution distances traveled. While this information helped bound many of the assumptions in

Table 4.5, the modeled scenarios result in an average estimate for a Mexico-wide emissions factor of weighted by market share data (e.g. in the baseline case, product distribution emissions from small plants are given a weight of 52% based on their market share).

We did not use actual operating data provided by interviewees in the model, however this was collected for nine plants. For example, while we collected electricity costs from interviewees, we based the water treatment electricity consumption needed for specific treatment processes (e.g. ozone) from Franklin & Associate data and other sources (Table 4.3). As such the interviewee data provides a useful comparison of emissions between the modeled scenarios and plant data from water source to the household. We estimated an empirical LCA for each of the nine plants that provided field data about actual electricity consumption in their plants, water use estimates, gasoline use, and distribution patterns. For values that the plants could not provide, literature values were used (e.g. plastic production emissions). The field data is limited and not representative for all of Mexico, and therefore serves as a snapshot.

4.2.7 *Back of the Envelope: Garrafón Emissions Compared with Tap Water and Household Treated Water*

We compare results of the modeled emissions for garrafones to emissions associated with tap water and tap water plus household water treatment (Reygadas *et al* 2014). For bottled water, tap water, and household-treated tap water, emissions estimated for the baseline case reflect cradle-to-use emissions and therefore the comparison does not include emissions associated with waste and disposal. We use estimates of Mexico's population and bottled water consumption volumes to estimate the emissions savings associated with switching between garrafones and tap water or household treated tap water. This back-of-the-envelope calculation represents the 'best case' scenario of savings potential, assuming households have access to tap water supply and switching from garrafones to treated tap water is a healthy and subsidized option.

4.3 Results

4.3.1 *Life cycle Assessment Emissions of Garrafones*

We estimated the average global warming potential of garrafón use for three different sets of assumptions to produce baseline, low, and high emissions scenarios across Mexico. Our baseline model scenario estimates garrafón water emissions of 44 kg CO_2eq/m^3, with a low emissions estimate of 21 kg CO_2eq/m^3 and a high emissions estimate of 75 kg CO_2eq/m^3. High and low scenarios can be interpreted as hypothetical minimum and maximum values given the assumptions in this LCA. Even the high emissions estimate is lower than the LCA conducted by Franklin & Associates for the same size bottle in the US (126 kg CO_2eq/m^3). This is largely due to the use-case in Mexico, where garrafones are frequently re-used and often last several years. However all scenarios are likely to underestimate the full life cycle emissions of garrafones, because the emissions associated with transportation from landfill and recycling at end-of-life are not included.

Figure 4.3 shows emissions for the three modeled scenarios. Notably, emissions associated with all phases of garrafón production except materials and garrafón distribution (Figure 4.3a) make up 7.4 kg CO_2eq/m^3 in the baseline case, compared with 36 kg CO_2eq/m^3 associated with materials and garrafón distribution (Figure 4.3b).

The phases of plastic resin production, bottle fabrication, and cap production collective make up bottle production emissions. Collectively we model that bottle production contributes 12% of life cycle garrafón emissions, or 5.1 kg CO_2eq/m^3 in the baseline case (Low: 2.2 kg CO_2eq/m^3; High: 12.2 kg CO_2eq/m^3). Polypropylene caps contribute a high percentage of modeled emissions (4.8%) relative to bottle fabrication (0.7%) because they are not reused and thus 100% of the emissions of cap production are allocated to each garrafón. LCAs of smaller bottles (as reviewed in Table 4.1) consistently identify plastic bottle fabrication as the phase with the largest energy requirements (Gleick and Cooley 2009, Jungbluth 2005, Quantis 2010). For example, as a percentage of the life cycle of energy use, Gleick and Cooley 2009 estimate that the three phases that make up bottle production for disposable bottles account for 39% to 71% of energy intensity, depending on model assumptions (Gleick and Cooley 2009). For comparison, we tested the contribution of bottle production in our model when by assuming that PET and PC bottles are used only once, and we find that bottle production emissions account for 74% of total estimated emissions in this case.

Emissions associated with bottling plant operations include water sourcing (i.e. municipal, trucked, or well water), water treatment at the plant, bottle filling, and washing. While source water (i.e. municipal, trucked, or well water), has a relatively low emissions contribution overall (1.0 kg CO_2eq/m^3 in the baseline case), water that is pumped and transported by trucks drive the majority of water source emissions in all modeled scenarios (Figure 4.4). Water treatment emissions intensities are similarly low in the baseline case (0.84 kg CO_2eq/m^3) and across model scenarios (Low: 0.3 kg CO_2eq/m^3; High: 1.3 kg CO_2eq/m^3). Emissions from bottle filling and washing are minimal. Collectively, bottling plant operations contribute 5.1% to overall emissions modeled in the baseline case, or approximately 2.3 kg CO_2eq/m^3 (Low: 1.1 kg CO_2eq/m^3; High: 4.8 kg CO_2eq/m^3).

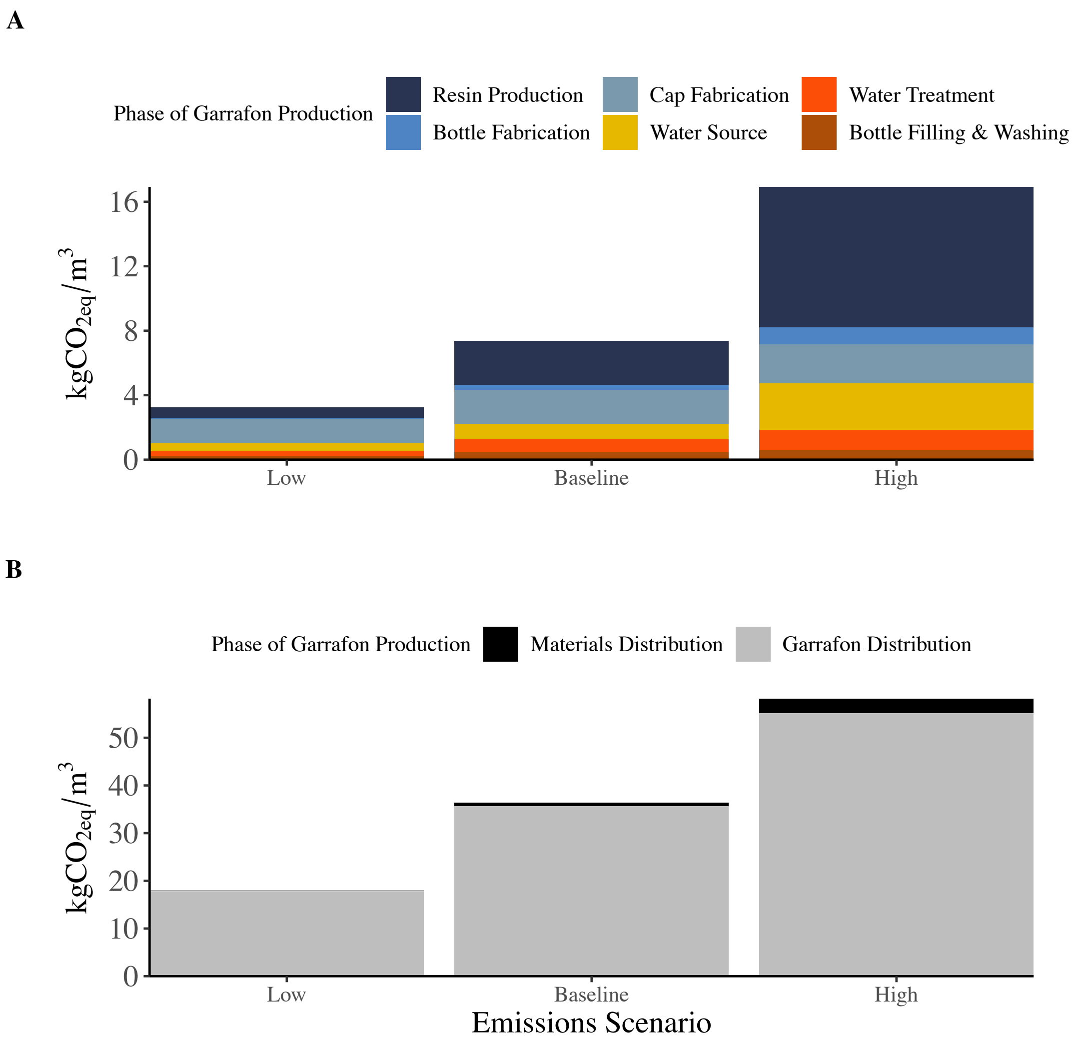

Figure 4.3 Greenhouse gas emissions of garrafón production across three emissions scenarios.

A shows all phases of garrafón production excluding distribution phase, which shows the largest proportion of emissions and therefore is shown separately in plot **B** (note the difference in y-axis). Materials distribution refers to transportation associated with plastic resin to the bottling company or treatment plant, a phase which is divided by the number of times a bottle is reused. Garrafón distribution refers to transportation associated with the product distribution route from treatment/refill plant to the home.

Materials and product distribution account for 83% of the overall emissions (36 kg CO_2eq/m^3) in the baseline case–but most of these emissions are associated with garrafón distribution from plants to homes and back. The Beverage Industry Roundtable found that distribution accounted for 26% to 64% of bottle emissions in Europe and 18% to 53% of bottle emissions in North America for 1.5L and 500mL bottles (Beverage Industry Environmental Roundtable 2012). The high variability of distribution in these models reflects varying assumptions and analysis boundaries. However, they also indicate relatively high uncertainty in estimating distribution routes and reflect sensitivity of emissions to transportation estimates. In the low and high emissions scenarios modeled here, distribution ranges from 18 kg CO_2eq/m^3 to 58 kg CO_2eq/m^3. Distribution is a lower percentage of overall garrafón emissions in the high-emissions scenario (75%) because we assume lower bottle reuse and thus plastic resin becomes a greater driver of emissions.

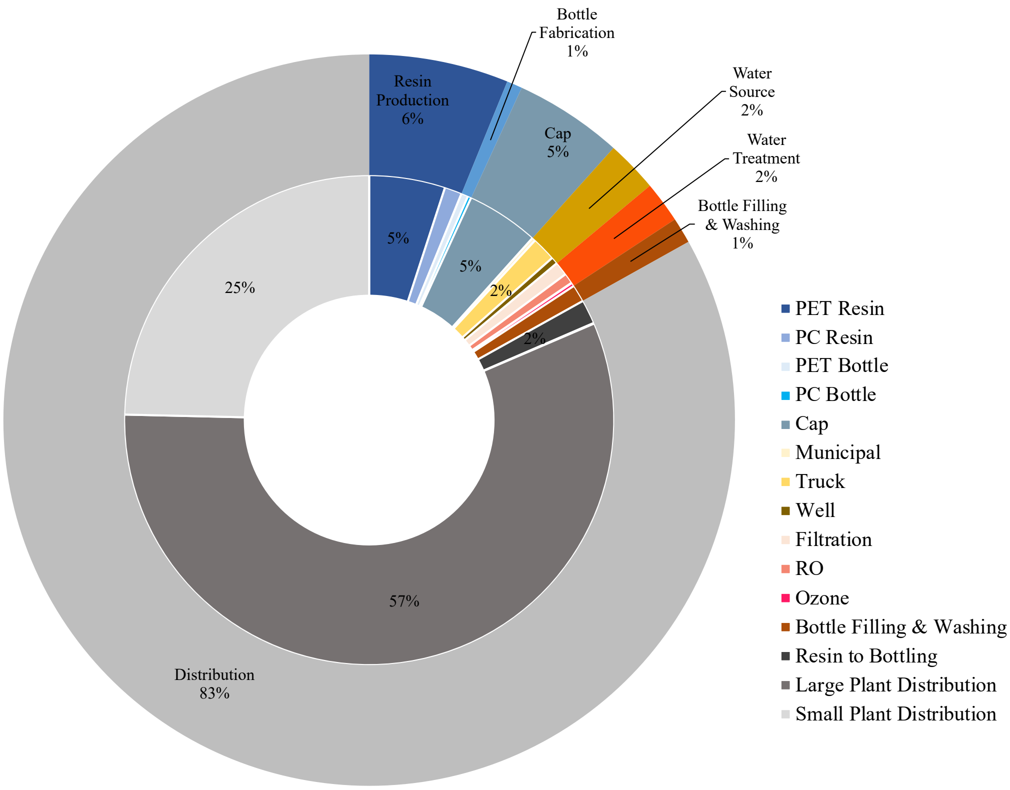

Figure 4.4 Baseline modeled parameters contribution to estimated garrafón emissions.

Outer ring of donut chart shows the overall phase of production and the percentage emissions contributed from each phase. Inner ring shows the sub-processes associated with each phase, and their contribution to overall emissions. Sub-processes that contribute less than 0.5% are not shown (i.e. emissions from UV treatment and emissions from material distribution of bottles to bottling plant). Labels are shown for processes that contribute at least 1% to overall emissions. PET = Polyethylene terephthalate; PC = Polycarbonate; RO = Reverse Osmosis.

4.3.2 *Nine LCAs based on Field Data*

We estimated LCA emissions of garrafones based on interview data using actual water treatment plant electricity use, distribution distances and gas-use, as well as re-use rates shared with us by nine interviewees (five based in Chiapas and four based in Baja California del Sur). For modeling the cradle-to-use emissions of these field data we used emissions data associated with bottle production (resin production, plastic fabrication, and materials distribution) from the baseline scenario model. The average estimated emissions per garrafón from the field data was about 25% lower than the baseline model emissions, at 33 kg CO_2eq/m^3 garrafón, with an interquartile range of

18 to 47 kg CO_2eq/m^3 garrafón (Figure 4.5). When we weighted emissions by garrafón production volume across the 9 field-based models, the average decreased to 30 kg CO_2eq/m^3.

Based on electricity cost data provided by bottling plants, we estimated monthly electricity consumption and applied the emissions factor used for the baseline model (0.57 kg CO_2eq/m^3). We calculated average weighted emissions from source water across the nine plants as 1.1 kg CO_2eq/m^3 and water treatment were estimated to be 2.5 kg CO_2eq/m^3 using the baseline electricity factor (Low: 1.8 kg CO_2eq/m^3; High: 4.2 kg CO_2eq/m^3). These findings are well within range of the modeled emissions.

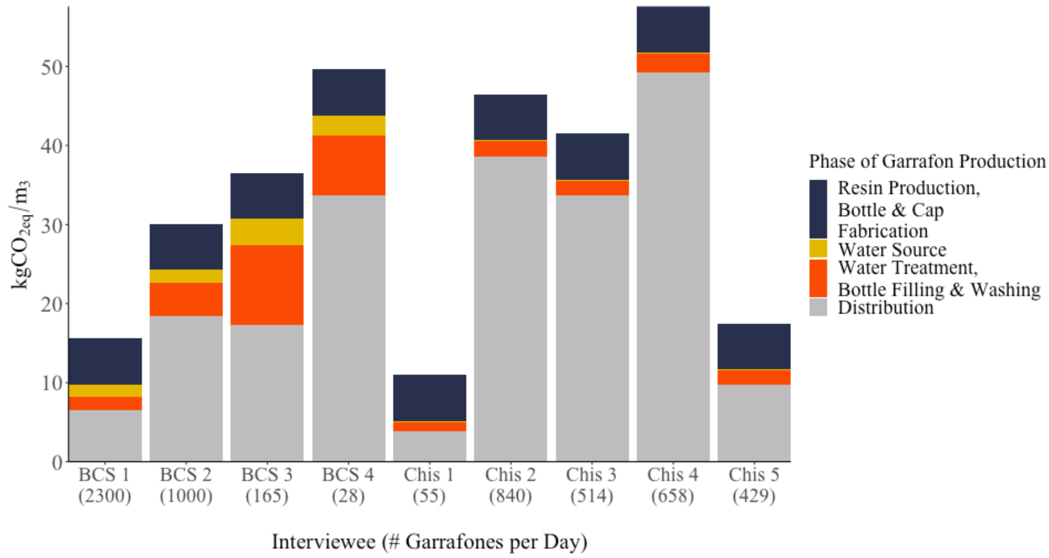

Figure 4.5 Field data results of emissions for bottled water in México.
SCC = San Cristobal de las Casas in Chiapas; BCS = Baja California Sur. Number of garrafones distributed per day in parentheses. Bottle and cap fabrication is the same across all estimates, based on the baseline modeled estimate.

From Figure 4.5 we note that distribution makes up the majority of overall emissions, but that there is substantial variation within and across the two regions. Interviewee Chis 1 was a refill station and had no reported product distribution routes and therefore we used the estimate of emissions associated with customer car pick up and a 50% travel allocation. Baja California Sur interviewees had higher average water treatment, bottle filling, and washing associated emissions (5.9 kg CO_2eq/m^3) compared with those in Chiapas (1.8 kg CO_2eq/m^3) due to higher electricity use. This is consistent with the high levels of salinity in Baja and the presence of more reverse osmosis treatment. Emissions associated with garrafón distribution averaged 23 kg CO_2eq/m^3, but plants in Chiapas had higher average distribution emissions (27 kg CO_2eq/m^3) compared with the Baja interviewees (19 kg CO_2eq/m^3).

4.3.3 *Back-of-the-Envelope: Emissions Reductions Potential*

Baseline emissions estimated in the Mexico-wide LCA model are 168 times greater than municipal tap water (0.26 kg CO_2eq/m^3) and 46 times greater than emissions associated with household-treated tap water (0.95 kg CO_2eq/m^3) (Reygadas et. al 2014). Switching households from garrafón use to tap water or treated tap water would thus reduce drinking water associated emissions by 99% and 98%, respectively.

To estimate emissions associated with country-wide garrafón use we need estimates of garrafón consumption volumes country wide. We do this using two back-of-the-envelope approaches. For the first estimate, we use bottled water sales volume estimates. Garrafón consumption makes up an estimated 2/3 of the volume of total bottled water consumption in Mexico (Rodwan 2017), which was estimated at 35.8 million cubic meters in 2019 for at-home consumption (Statistica n.d.). If we assume two-thirds of annual consumption of bottled water in 2019, this results in an estimate of 23.9 million cubic meters of drinking water per year consumed by garrafones. For the second estimate, we use data on reported rates of garrafón use at home and average water consumption for dinking water. An estimated 76% of Mexico's roughly 129 million person population use garrafones or bottled water as their primary source of drinking water (INEGI 2017). It is likely that garrafones are more common as a primary household source. For this exercise, we use the INEGI statistic of 76% and assume 100% of household bottled water use for primary drinking water comes from garrafones. The implications of this assumption are an underestimation of emissions associated with household bottled water use, because smaller disposable plastic bottles have higher emissions than garrafones. If we assume 3 liters of drinking water per person per day (recognizing a range from 2-4 based on health recommendations) for 76% of the population of Mexico, this results in 107 million cubic meters of drinking water per year consumed by garrafones.

Notably the second estimate of garrafón water consumption levels based on INEGI statistics is much higher than the estimate based on recorded bottled water sales. This could reflect an increase in garrafón sales relative to overall water consumption since 2017, but it could also reflect the fact that there are thousands of refill stations and purification plants that are unlikely to be captured in high level market research statistics. We can therefore estimate emissions associated with garrafón use for the range of 23.9-107 million cubic meters per year using the emissions factor of 0.57 kg CO_{2eq}/ m^3.

Using the baseline emissions factor for garrafones assuming low (23.9 million m^3) and high (107 million m^3) consumption estimates, country-wide garrafones emissions are modeled to be 1 to 4.7 million tonnes of CO_{2eq} per year, respectively (Table 4.6). While this is merely a back-of-the-envelope estimate subject to high uncertainty, the results in Table 4.6 suggest that garrafón emissions are on the scale of 10^6 tonnes of CO_{2eq} per year. These results are likely an underestimation given that we do not estimate the emissions associated with plastic disposal and recycling or consider PET resin imports from distances beyond North America. Equivalent emissions from tap water would be approximately 6,200 - 28,000 tonnes of CO_{2eq} per year, and with household-treated tap water slightly higher at 23,300 - 102,000 tonnes of CO_{2eq} per year (Table 4.6).

Table 4.6 Modeled annual garrafón emissions in Mexico compared with tap water and household water treatment emissions.

Estimated Annual Garrafón Consumption	Baseline Garrafón Emissions	Tap Water	Tap Water + Household Water Treatment
Million m^3 per year	Million tonnes CO_{2eq}/year		
23.9	0.94	0.006	0.020
107	4.7	0.028	0.102

4.4 DISCUSSION

Few studies have conducted life cycle assessments of bottled water outside of the U.S. and Europe, but six of the top ten countries with the highest bottled water consumption are low or middle income countries (Cohen and Ray 2018). This study represents one of the first life cycle assessments of home and office delivery bottles (known as garrafones in Mexico) specifically aiming to capture bottle use and distribution patterns emerging in low and middle income countries. We conducted a process-based LCA of 20 liter garrafones augmented by expert input through interviews and an empirical estimate of emissions associated with nine bottling plants. We used country-wide average emissions associated with electricity and municipal water supply, as well as various garrafón distribution routes, to model scenarios relevant to Mexico's bottled water market share.

Overall, we model a baseline emissions factor for garrafones to be 44 kg CO_2eq/m^3 (Low: 21; High: 75 kg CO_2eq/m^3). This is lower than emissions from single use plastic reported in the literature (where emissions factors are typically greater than 150 kg CO_2eq/m^3). However, compared with emissions associated with tap water or household treated water, garrafones have a substantial climate impact. High levels of bottle reuse in Mexico drive down the lifetime emissions associated with a single garrafón because the emissions from resin and bottle production are divided over the lifetime use. Consequently, emissions associated with garrafón distribution (from treatment plant/refill station to households) make up the majority of bottle emissions in the three model scenarios.

We also estimated an empirical life cycle assessment for nine bottling plants in Chiapas and Baja California Sur. Our model using empirical data suggests that regional differences in plant operations are an important consideration when evaluating intervention points for emissions reductions in source water, water treatment operations, and product distribution. We found higher average emissions associated with distribution in plants interviewed in Chiapas, but Chiapas had lower water source and treatment emissions than those plants in Baja. This is a small convenience sample and does not represent the case of Mexico, but rather serves as preliminary findings to suggest the need for more research on the potential importance of regional heterogeneity when considering ways to reduce garrafón emissions.

There are several gaps in our analysis. First, we only capture emissions associated with cradle-to-use phases of the garrafón life cycle. We do not estimate emissions associated with end-of-life recycling and disposal. Arguably, the energy intensity (transportation, disposal and handling) of recycling garrafones would increase the garrafón emissions relative to tap water (Nessi *et al* 2012) and household water treatment. Secondly, given limited data on the distribution, use, and accurate market-share of the bottled water industry in Mexico, our estimates are subject to uncertainty. Nonetheless, in combination with literature-values and our consultation with bottlers, plant manufacturers, and distributors in Baja California Sur and Chiapas, we believe our scenarios capture a wide range of likely emissions profiles for garrafones, and the variability in modeled product distribution emissions from low to high scenarios are similar in other LCA studies on single use bottled water. Given the relevance of distribution to model outcomes, more data is required to improve estimates in contexts where bottle reuse is high. This could be accomplished with a representative survey of bottlers and distributers. Finally, bottled water has many other environmental impacts beyond climate emissions that should be included in a full sustainability assessment. These include impacts such as plastic waste (Horowitz *et al* 2018), chemical leachates from plastics, and draw-down of groundwater resources.

Our findings provide empirical evidence that using bottled water to access (potentially) safe drinking water (SDG 6) is occurring at the expense of other core Sustainable Development goals like SDGs 11 (Sustainable cities and communities) and 12 (Responsible consumption and production). We hypothesize that this trend will continue because there is little confidence in publicly supplied tap water and private bottling companies have largely captured the market for household drinking water. Even where households have access to treated municipal tap water, households are less likely to drink tap water over bottled water. INEGI's national survey (2017) found that households use garrafones as a primary drinking water source because they think it is the healthier option (69% of survey respondents) or because they dislike like taste or color of public supply (19.6%). Households purchasing bottled water because they perceive it as healthier may see bottled water as a necessity (i.e. not a choice). Lack of clear tap water quality information from municipalities, combined with incentives for bottled water corporation, are hypothesized institutional drivers of persistent household bottled water use (Montero-Contreras 2016). Few studies investigate the link between perceived water quality and actual water quality, but a study in three Oaxacan communities found that while perception and actual water quality may vary, most households use bottled water anyway. In their sample, 100% of households in two of the communities and 84% in the third community relied on bottled water regardless of their perception of tap water as safe or unsafe (Rowles *et al* 2018). There is very little data available regarding the water quality of garrafones, especially off-brand small operations that have extremely low costs. This is particularly problematic because bottled water companies are not subject to the same oversight as public suppliers (Montero-Contreras 2017, 2016).

There are substantial equity considerations attending the unsustainable use of garrafones. The perceived safety of garrafones over tap water may drive households to pay for bottled water despite its high costs. Garrafones range in price from 10 to 45 MXN per bottle (Walmart 2019, Reddit 2015), with larger brands costing more. Comparatively, tap water for the same volume of water costs between 1.32 and 4.67 MXN across states in Mexico (average = 2.07 MXN; SD = 0.98 MXN) based on the national tap water database (IMPTA and SEMARNAT 2018). Rowels et al. 2018 used average income levels in three communities in Oaxaca and estimated that households spend between 2.4% - 4% per month on bottled water alone (Rowles *et al* 2018). This corroborates a study in the state of Chihuahua of nearly 400 households, where bottled water expenditures made up 3.6% of the median income levels for households in the study (Vásquez *et al* 2009). When combined with tap water costs, households at the median income level are spending 7.5% on water and bottled water combined. These cost burdens to households not only substantially exceed common thresholds for affordability, but they do not reflect how garrafón prices impact economically vulnerable households.

The relationship between affordability and water quality requires more investigation. Higher income households consume water from larger brands, which in general are subject to more oversight than small plants and tend to have better microbiological quality. Many small bottling plants do not follow best treatment practices and contamination rates are higher (based on communication about field work by (Reygadas 2014)). Consequentially, lower income households are potentially/likely exposed to unsafe water at a higher rate than the higher income population–all while spending more relative to tap water, which potentially results in un-affordability. While small treatment plants and refill stations have notably lower prices than name-brand garrafones, they are still more expensive than tap water and are not guaranteed to have safe water quality.

Our results support policy action to reinforce public water supply and reduce the use of garrafones as a primary drinking water source. Improving quality of piped systems could reduce garrafón consumption, thus leading to reduction in greenhouse gas emissions and better affordability for households in the long run. Mexico's residential building sector accounts for 26% of its national electricity use (14% of GHG emissions), and its demand is projected to grow at 5% per year (IADB 2012b). The residential sector is thus a critical target for aggressive greenhouse gas abatement, and prioritizing the sustainability of household drinking water could simultaneously address inequities by increasing the safety of affordable tap water.

Financing related to mitigation (low interest loans and/or funds) could meaningfully contribute to improving piped water services. Comparing the results of the garrafón model to previous work on the life cycle emissions of household water treatment in Mexico, we find that switching to household water treatment could yield emissions savings as much as 98% per cubic meter in the baseline case. Undoubtedly, investment in public water infrastructure will require substantial investment in communication to match marketing from the bottled water industry, where health-benefits and water safety are primary marketing talking points.

For sustainable transitions to work, policymakers need to evaluate trade-offs across and within SDGs (Pradhan et al. 2017) and develop policies that encourage decoupling of essential needs like drinking water from emissions intensive processes (Sachs et al. 2019). Our findings point to the need for better metrics to evaluate water access pathways in the context of SDGs, where water access may be realized through bottled water consumption at the cost of other goals, like sustainable production and consumption (SDG 12). Metrics to capture drinking water access from improved sources can be augmented with life cycle-assessments to compare the sustainability of different access pathways and benchmark country-wide trends in water access and sustainability.

CONCLUSION

Metrics and indicators play a key role in the representation of water access problems and their proposed policy solutions. By comprehensively reviewing metrics for water affordability (Chapter 2), developing new metrics and empirically investigating them and their equity implications in the case of California (Chapter 3), my work advances the field of water affordability for human right to water policies. In Chapter 4, I turn to the question of household drinking water sustainability in low and middle income countries, using Mexico as a case study. While these projects focus on different dimensions of household water access, there are lessons emergent from both projects individually and their implications when considered together. Below, I summarize the key findings from water affordability research, followed by implications for research and implications for policy. I then focus on the findings and implications for the bottled water research, followed by a reflection on both of these projects.

WATER AFFORDABILITY KEY FINDINGS

Water affordability has traditionally been measured in the U.S. through a two-pronged framework evaluating the impact of water bills on median income levels, alongside measures of financial capability of water systems to cover their costs (US EPA 1998a). This approach has received criticism for inadequately capturing low-income affordability struggles (Davis and Teodoro 2014). EPA commissioned evaluations of this approach in 2002 (EPA Science Advisory Board 2002) and in 2017 (NAPA 2017). The most recent review makes a strong case for a focus on equity. NAPA advised the EPA to begin measuring affordability for low-income households and considering social-economic vulnerability of community water systems. As of the writing of this dissertation, however, no formal revision of the federal criteria has been adopted. State level policy has advanced this conversation forward, with California leading efforts to institutionalize the human right to water. The human right to water and the broader Sustainable Development Goals provide strong aspirations for social equity, non-discrimination, and sustainability. These frameworks do not, however, provide insight into how to measure and track water access at the local level.

Since 2015, I have worked with California's Office of Environmental Health Hazard Assessment to develop metrics for water affordability. This context afforded me incredible insight into the process of developing metrics for the first human right to water tracking tool in the country. It became clear when I began that affordability–as a normative and highly contested term–would never be "defined" but rather discussed, debated, and measured over time. Chapters 2 and 3 of this dissertation contribute to this dialogue, offering recommendations for best practices and analytical insight into water affordability challenges in California.

Chapter 2 contributes to this discussion with a critical literature review on definitions and measures of water affordability. Several measures exist, though affordability ratios originally developed by EPA (Mack and Wrase 2017) are most prominent. A small subset of studies promote methods like the residual income approach (Miniaci *et al* 2008) which has also been used in the housing affordability literature. A primary finding of this research is that while many researchers use the same overall measure (the affordability ratio), few studies are clear about the normative assumptions underlying the inputs to their metrics.

We also identified several areas of ongoing discussion about how to measure water affordability, which include: the spatial and temporal scale and scope of study, water to meet basic needs, the multiple costs facing households, available income, and the criteria for affordability. These themes reflect conceptual and pragmatic issues that researchers face when applying measures. I then evaluate the human right to water and SDG goals for social equity, non-discrimination and ecological sustainability to identify if and how measures incorporate these aims. This critical assessment resulted in a set of recommendations for researchers to transparently identify how the assumptions and parameters they use to measure water affordability might better incorporate sustainable development and human right to water aspirations.

In Chapter 3, I draw insight from the critical review of key themes from Chapter 2 to develop an approach for evaluating water affordability in community water systems as part of California's first human right to water tracking tool in the U.S. My research on water affordability is part of a broader set of human right metrics including water quality and access (Balazs *et al* 2019). I created metrics at the water system scale to evaluate affordability essential needs water bills at median, poverty, and deep poverty income levels. This resulted in three new affordability ratios that have been advanced as metrics for monitoring affordability in California's community water systems. These metrics innovate on previous affordability research by developing multiple ratios within each water system and focusing on the distribution of outcomes, thereby avoiding a binary determination of affordable / unaffordable.

A key barrier to measuring water affordability emerged in Chapter 3: there is a substantial lack of comprehensive data at the same scale of analysis. For researchers and policy makers to advance water affordability metrics, information on water costs, incomes, and rate structures are required. Moreover, this data needs to be at a common scale for analysis. Before highlighting our findings, I summarize our approach to managing incomplete data because this is an important area for further research and implicates policy options (discussed in the next sections).

First, to gather data at the same scale for analysis, I compiled data from Census blocks, block groups, water system reporting systems, and water system boundaries. Through geographic manipulation and weighting, I built on previous work to incorporate all relevant affordability data to water system boundaries (Balazs *et al* 2012). While there are inherent errors in this approach (e.g. assuming that all households are homogenously distributed within block groups), there are few alternative methods. Advancing the assessment of social-demographic characteristics of households in water systems is an area ripe for future research.

Secondly, we found that many water systems had missing water bill data. Community water systems report water bills through the state's electronic Annual Reporting system, but only 52% of water systems in the state reported bills in 2015 (the data source used in this study). The smallest water systems had the most missing data, with an 18.5% reduction in representation of very small systems between the full sample (where very small systems make up 62.5% of systems) and the sample (where very small systems make up 44% of the systems). This reflects a moderate bias against representation of very small systems. To address this, I modeled missingness by a variety of water system characteristics (See Appendix B5 and B6) to identify potential confounders of missing data. These were then included in a generalized regression model for the different affordability ratios in the study sample to mitigate bias due to missingness in our outcome variable. This allowed us to estimate adjusted mean affordability ratios by system size. While this approach likely improves our estimates, the trends remained stable across the sample and the full population list.

Finally, we investigated the reliability of water bill data and the Census data used to estimate affordability ratios within water system boundaries and tested the impact of unreliable data on the study. The electronic Annual Report has not undergone an assessment for reliability, but several water systems had very high and very low water bills (range = \$5 to \$466 per month across systems). We investigated very low and very high water bills through our own survey and by excluding potential outliers in a sensitivity analysis. Most studies using Census social and demographic data do not investigate sampling error, but reliability of these estimates varies widely across region, with rural areas having less reliable data. This presents a problem for inference. To address this, we labeled water systems as having reliable or unreliable data where water systems overlapped with one block group. We then conducted our analysis with and without unreliable data points. Our results were robust to sensitivity analyses excluding potentially unreliable water bill and income data, which enabled us to confidently interpret the overall trends identified in the results.

Addressing these challenges in data quality and availability enabled me to analyze water affordability across water systems. I found that households earning median household income levels had relatively affordable water bills for essential needs water–making up 1% of median incomes on average across water systems. Affordability is regressive, with water bills for essential needs impacting county poverty and deep poverty level incomes substantially. The median county poverty level affordability ratio was relatively affordable (1.8%), but the impact of water bills is substantial for households earning at the deep poverty level. A quarter of water systems have bills that are 2.7% of county poverty level incomes and 5.3% of deep poverty levels. For economically vulnerable households in these systems, even average water bills (\$52.44 per month for 6 HCF) can be unaffordable.

Another key finding is that system size has an important role in affordability outcomes. Households served by small and very small systems (combined serving less than 3,300 people) have significantly higher affordability ratios across all three income levels than households served by large systems. Adjusting for measured confounders, I estimate affordability ratios at median income levels for very small system are 1.1% on average, and affordability ratios at county poverty and deep poverty income levels are estimated to be 2.2% and 4.3%, respectively. Affordability ratios for large systems at median, county poverty, and deep poverty income levels are calculated as 0.6%, 1.3%, and 2.7%, respectively. These findings could indicate diseconomies of scale, wherein smaller water systems have a smaller revenue base to distribute cost of provision.

While average ratios are higher (i.e. more unaffordable) in very small and small systems, the percentage of households in poverty is not substantially different across water system size. Therefore, in absolute terms, large water systems have a higher number of people in poverty. For these households, water bill burdens appear moderate, but they are near commonly used thresholds (e.g. 1.5% for economically disadvantaged communities) and reflect a lower bound estimate of water bill burdens. As with all estimates in this analysis, sewer and sanitation are not incorporated into overall costs. Future work should aim to incorporate these costs to understand the full affordability challenge to households.

Water Affordability Research Implications

Of the recommendations we identified in the critical review in Chapter 2, there are three that demand priority attention for the research community. First, affordability studies that disaggregate measures to reflect impacts on economically vulnerable groups are growing and this is promising. Chapter 3 contributed to this effort by looking at poverty and deep poverty levels within water systems. This enabled us to see the regressive impact of water bills at essential needs levels, and indicate affordability challenges even in large systems with relatively low water bills. But more work is needed to identify water affordability for historically marginalized groups. Few studies have evaluated how water affordability impacts communities of color, despite persistent environmental injustices for these communities across the U.S. Moreover, more research is needed to include affordability considerations for people not necessarily served by water systems, such as those relying on domestic wells, people living without homes, mobile home communities, and unincorporated communities.

A second recommendation is that more theoretical and case-study work is needed to identify the ways that water bills and costs impact household incomes. Most studies identified in Chapter 2 measure affordability at scales larger than the household level. Our study of community water systems in Chapter 3 has the advantage that the scale of analysis aligns with the holistic study of water access in California's human right to water project. However, neither this study nor any studies identified in Chapter 2 identify how households change their spending in response to unaffordable bills. We know very little about the internal structure of household budgets, and to what extent households paying for water are reducing spending on other essential needs (See: Rockowitz *et al* 2018, Cory and Taylor 2017). This implicates how we interpret water affordability measures. For example, affordability ratios that fall below some percentage of income, even after removing essential expenditures, may be seen as affordable. It is important to improve our understanding of how bills spending choices so that measures can be improved to reflect the way that water bills burden households.

Finally, affordability measures should be evaluated holistically alongside other aspects of water access, like physical accessibility and water quality. Similarly, households with poor water quality or high distrust in tap water will turn to bottled water (Javidi and Pierce 2018, Allaire *et al* 2019), which is more expensive with greater environmental impacts, as I show in Chapter 4. Communities identifying as African American, Hispanic, and foreign born have been shown to prefer bottled water over tap water in several studies in the U.S. (summarized in (Javidi and Pierce 2018)), leading to the hypothesis that there are potentially racial/ethnic inequities in affordability. Both of these examples identify areas where one dimension of water access–i.e. physical access as far as quality of pipes, or water quality–could negatively impact affordability while also having worse environmental outcomes.

Affordability measures in such a context can incorporate coping costs (Nastiti *et al* 2017b, Moore *et al* 2011) to more fully represent costs to households.

WATER AFFORDABILITY POLICY IMPLICATIONS

Water affordability surfaced to national consciousness when households were denied water access after mass water shut-offs across Detroit (Kurth 2017) and Baltimore (Colton 2017). However, water affordability programs across the U.S. are fragmented because there are no federal, and few state-level programs to ensure that people can afford their drinking water. Reporting on the state of the human right to water in the U.S., the United Nations Special Rapporteur on the right to water and sanitation found that from 1991-2011, one trillion dollars were spent on drinking water and wastewater supply and treatment, and that customers covered 90% of these costs (de Albuquerque 2011). These costs are growing in response to aging infrastructure and climate pressures (National Academy of Sciences 2019). The gap between rising costs, rising unaffordability, and disparate programs to address the challenge for low-income households is deeply concerning.

California is currently seeking to fill this gap within its borders. The State is developing a low-income rate assistance program to assist low-income households with water rates (State Water Resources Control Board 2019b). More recently, the State Water Board is conducting a Needs Assessment to assess broader water access needs including affordability (State Water Resources Control Board 2019a). The California Public Utilities Commission is evaluating measures to investigate affordability in their systems (CPUC 2019). All of these efforts are underway and reflect a promising shift to a focus on water affordability for lower-income households through state-led effort. Our research findings in Chapter 3 indicate that economically vulnerable households are cause for concern across all system sizes, because water rates, especially in urban areas, have been rising faster than inflation (Hanak *et al* 2014) while incomes have stagnated or declined for middle and low-wage earners (Reidenbach 2015, Gold 2015).

Despite growth in institutional support and funding to feasibly implement the right to water, there are still policy hurdles. Policy efforts require better data to effectively implement programs. We encountered substantial data limitations in accessing information for measuring affordability. Even where data were available, reliability is a persistent concern. Leveraging sensitivity analyses can provide insight into broad picture trends in academic research, but this may not be useful for monitoring of individual water systems. The low-income rate assistance proposal currently aims to support households directly, which may serve households better in the near term, though it is unclear how the state will identify low-income households. Currently proposals revolve around the concept of 200% below the Federal Poverty Line. This measure could be improved by using a California specific measures (e.g. 200% below the County Poverty Threshold) to reflect differences across California's counties. Because households in poverty or deep poverty may face significant affordability challenges regardless of system size, a program like LIRA that goes beyond the traditional focus on small systems is likely to be more comprehensive.

At the same time, efforts are needed to encourage data collection among small water systems. Our findings in Chapter 3 indicate higher bills and worse affordability outcomes (i.e. more unaffordable bills) for very small water systems. This corroborates the broader literature on small water systems, where disparities in governance, technical, managerial and financial capacity, and state investment results in inequitable water access provision especially in small systems (McFarlane and Harris 2018, Balazs and Ray 2014). Improving information access for monitoring in the human right to water requires better data collection from these systems, but not in a way that exacerbates existing burdens. A promising direction could be leveraging purposive random sampling to develop representative samples of water access data across small systems annually.

BOTTLED WATER IN MEXICO KEY FINDINGS

In Chapter 4, I turn to the question of household water access sustainability. People in the U.S. and internationally turn to bottled water both when water quality is known to be a risk (Allaire *et al* 2019, Pattanayak *et al* 2005, Laughland *et al* 1993, Walter *et al* 2017) and when people perceive their water to be a risk (Pierce and Gonzalez 2017a, Doria *et al* 2009). In Mexico, bottled water use is the highest per capita globally, with an estimated two-thirds of bottled water consumption in the form of 20 liter bottles, known as garrafones (Rodwan 2017). At home, 76% of households relying on bottled water or garrafones as their primary drinking water source (INEGI 2017). Through a contract project with the Inter-American Development Bank, I worked with other graduate students at ERG to develop metrics of greenhouse gas emissions associated with residential water use in Mexico. Our metrics helped develop Mexico's first sustainable and affordable housing initative under the country's national climate mitigation plan.

My approach was to model the global warming potential impacts of an average garrafón over its lifecycle using an input-output model for the energy requirements over seven stages of garrafón production (from production of the plastic to use at home). I drew on expert interviews to parameterize the model, for a baseline, low, and high emissions scenario. These metrics reflect a nation-wide representative metric for the emissions associated with garrafones. In the baseline case, we model a baseline emissions factor for garrafones to be 44 kg CO_2eq/m^3 (Low: 21; High: 75 kg CO_2eq/m^3). Garrafones are thus 46 times the global warming potential of tap water treated by a household water treatment system and 168 times more than municipal tap water. Our findings in Chapter 4 suggest that garrafones have lower emissions over their lifetime compared with similar sized bottles evaluated in the U.S. (an estimated 126 kg CO_2eq/m^3); however, as we did not model emissions associated with end-of-life it is likely that our model slightly underestimates emissions (Franklin Associates 2009b).

BOTTLED WATER IN MEXICO RESEARCH IMPLICATIONS

A majority of studies investigating the climate impacts of bottled water take place in Europe (Fantin *et al* 2014) or the U.S. (Gleick and Cooley 2009). However most bottled water use is happening outside of these countries. Chapter 4 introduces the first study on the emissions associated with garrafones, which are a drinking water staple in Mexico and a growing water source, globally. Mexico is a good proxy for household bottled water use patterns in low and middle income countries, where bottle reuse is high and the market share of micro-to small-scale purification and bottling plants is high. More research is needed internationally to understand the extent of bottled water related emissions and other environmental impacts (e.g. plastic waste).

As with the affordability research in California, data is relatively sparse on patterns of household water access in Mexico. The model I developed is novel in its use of interviews and compilation of disparate datasets in order to model the case in Mexico. This study emphasizes the importance of an interdisciplinary approach when data-constraints are substantial. Data constraints produce some uncertainty in our approach, and research to test and improve on LCAs in data-constrained environments are needed to minimize uncertainty and costs. For example, our expert interviews were reflective of a range of potential emissions scenarios in Mexico, but they were not representative of the country as a whole. A random and representative survey combined with LCA to bound assumptions are promising methods for future studies. There is much to learn from future work investigating regional emissions scenarios or expanding this research to new countries. Moreover, future work should aim to build on existing models and incorporate other waste streams like landfill emissions and plastic waste (Dettore 2009, Nessi *et al* 2012) in Mexico and other low and middle income countries.

BOTTLED WATER IN MEXICO POLICY IMPLICATIONS

Bottled water as a primary water access source is legitimized by the current SDG framework because it is considered an improved source. Where no safe water access from more traditional sources exist, this may be a temporary necessity. However, it is not a pathway to sustainable water access (Cohen and Ray 2018). Decoupling of carbon emissions from consumption in society is a priority goal of the SDGs (Sachs *et al* 2019). This is especially relevant for essential needs like water, food, and shelter. I use the life cycle approach as a tool for developing metrics that capture this challenge for household water use.

This project has had immediate policy implications. The bottled water emissions metric was part of a broader set of metrics for emissions associated with residential water use (Reygadas *et al* 2014), household water treatment, and tap water in Mexico. These were used to advance green financing and mortgages for low-impact houses in Mexico in the Ecocasa Program (IADB 2013). The Ecocasa program has been praised as an example of building housing that is both affordable and sustainable (Annan 2013, Demidchuk *et al* 2018). Ecocasa houses emit about 20% less than conventional homes and the full program aimed to build 27,600 sustainable homes by 2019 to save 1 million tonnes of CO_2 emissions across all emission sources within the homes (IADB 2013, Annan 2013).

There are several broader implications of this work. Policies aiming to improve water access need to account for the sustainability of providing the human right to water. Ideally, investment in infrastructure and development of trust between households and water providers would displace the need for bottled water in Mexico and beyond. Nearer term interventions could include a switch to household treated tap water where households have access to piped supply. Emissions associated with household treated tap water are modeled to be 46 times lower than with garrafones, indicating that emissions reductions would be substantial. A switch to household treated tap water would require that household's comply with water safety protocols to ensure high water quality (Reygadas 2014), a task that is non-trivial. Within the bottled water network itself, the re-use patterns of bottles in Mexico drive down life cycle emissions associated with plastic resin. This results in distribution as the biggest contributor to garrafón emissions in the baseline model and indicates a key starting point for investigating possible interventions. Improving the sustainability of transportation and distribution of garrafones could contribute to further reductions in areas where bottled water is a necessary interim solution to inadequate water access.

REFLECTIONS ON EQUITY & SUSTAINABILITY OF HOUSEHOLD WATER ACCESS

A key theme of this dissertation is the role of social equity in developing and analyzing metrics for water access. Historically, social equity and public health goals had precedence over economic efficiency in the water industry (Bakker 2001). In the last four decades, however, utilities and water providers have been equally if not more focused on cost-recovery, or economic efficiency in the U.S. and internationally. The human right to water has sought to intervene in this discourse with an assertion that people have an ethical claim to safe, affordable, and accessible water. By focusing on economically vulnerable groups in Chapter 3, the metrics I propose offer insight into affordability challenges facing economically vulnerable groups within California water systems–especially in small water systems.

Another theme explored in these projects is the role of household water access choices. In Chapter 2, I identified that most affordability measures consider direct costs of water bills to households, but in areas where water quality is inadequate, coping costs like bottled water are necessary to include in analyses. In addition to creating a social equity concern about increased costs for households, bottled water has significant climate impacts. In Chapter 4, I find that these impacts may be less per use when households use larger, reusable bottles–but compared with tap water the global warming potential is substantially higher. In pursuit of SDG 6–safe and affordable water for all–we cannot lose sight of the impacts of our technological solutions.

Residential water use is less intensive–from a consumption standpoint– than other water use sectors like agriculture. However, consumption has received substantial attention as the focal point for household water access sustainability in the SDGs (which have a target on water efficiency) and in academic literature. Bakker et al. 2010 argues that a pathway toward social equity and ecological sustainability is to focus on "ecological governance" wherein water bills are tiered to minimize waste and recover costs, but protective of essential use water for lower-income households, sustainability and equity goals may be jointly driven (Bakker 2010). The focus on minimization of waste is important. Yet in contexts where water access is low or perceptions of tap water quality are low, this will not be an adequate response. Bottled water use is becoming a permanent installation in several countries like Mexico, but also in the U.S. To counter this trend, substantial public investment is required to upgrade the failing infrastructure and build more sustainable and reliable systems where access is limited. Perhaps even more importantly, water systems–whether centralized or decentralized–need to build trust with households to counter bottled water use.

Future research is needed to build on the metrics developed in this dissertation. Studies could incorporate bottled water and replacement costs into affordability ratios where water quality is poor or communities regularly drink bottled water. Research into interactions among SDGs might benefit from metrics capturing the emissions impacts of bottled water use. Further, they can provide nuance to our representation of water access successes by underscoring areas where water access is obtained at an environmental cost. Finally, while recent scholars in sustainability science urge researchers to focus on the synergistic or antagonistic interactions between SDGs (such as SDG 12 for sustainable consumption and SDG 6 for safe water access), there is less discussion about potential causal mechanisms relating different SDG areas. Studies might seek to investigate whether or not the "equality/sustainability hypothesis" pertains to global patterns in bottled water use and affordability. The equality/sustainability hypothesis posits that social and economic inequality leads to higher pollution and environmental degradation (Cushing *et al* 2015). On the global scale, six of the top ten bottled water consuming countries are considered low or middle income countries (Cohen and Ray 2018). Is global inequality among and within countries driving unsustainable, and potentially unaffordable use?

With improved data collection, attention to cross-links between water affordability, sustainability, and other water access measures like quality, these questions can be asked. This dissertation offers an initial set of measures to improve water access monitoring for affordability and the sustainability of household water use. Such metrics are increasingly important to ensure representation of water access barriers for people and communities in a world that is growing more unequal year on year.

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APPENDIX A

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APPENDIX B

B1 DATA SELECTION & CLEANING OVERVIEW

Systems from the State Water Board's electronic Annual Review database from 2015 were included in the study if they met the following criteria:

1) Water system reports monthly water bill at 6 hundred cubic feet *and*,
2) Water system has no reported monthly water bill at 6 hundred cubic feet but reported a Flat Base Rate billing structure:
    a) *and* a monthly billing frequency *or* an explicit billing frequency in notes (e.g. "per quarter")
    b) *and* no other prices were reported in non-FBR rate categories.

Systems were excluded from the study if they met the following criteria:

1) Water system was not in community water system list obtained from the Office of Environmental Health Hazard Assessment (n = 2,901 community water systems);
2) Water system reported rates instead of monthly water cost, demonstrated by looking at a system's reported rate structure and reported rate prices;
3) Water system had clear reporting error;
4) Water system had no median household income data.

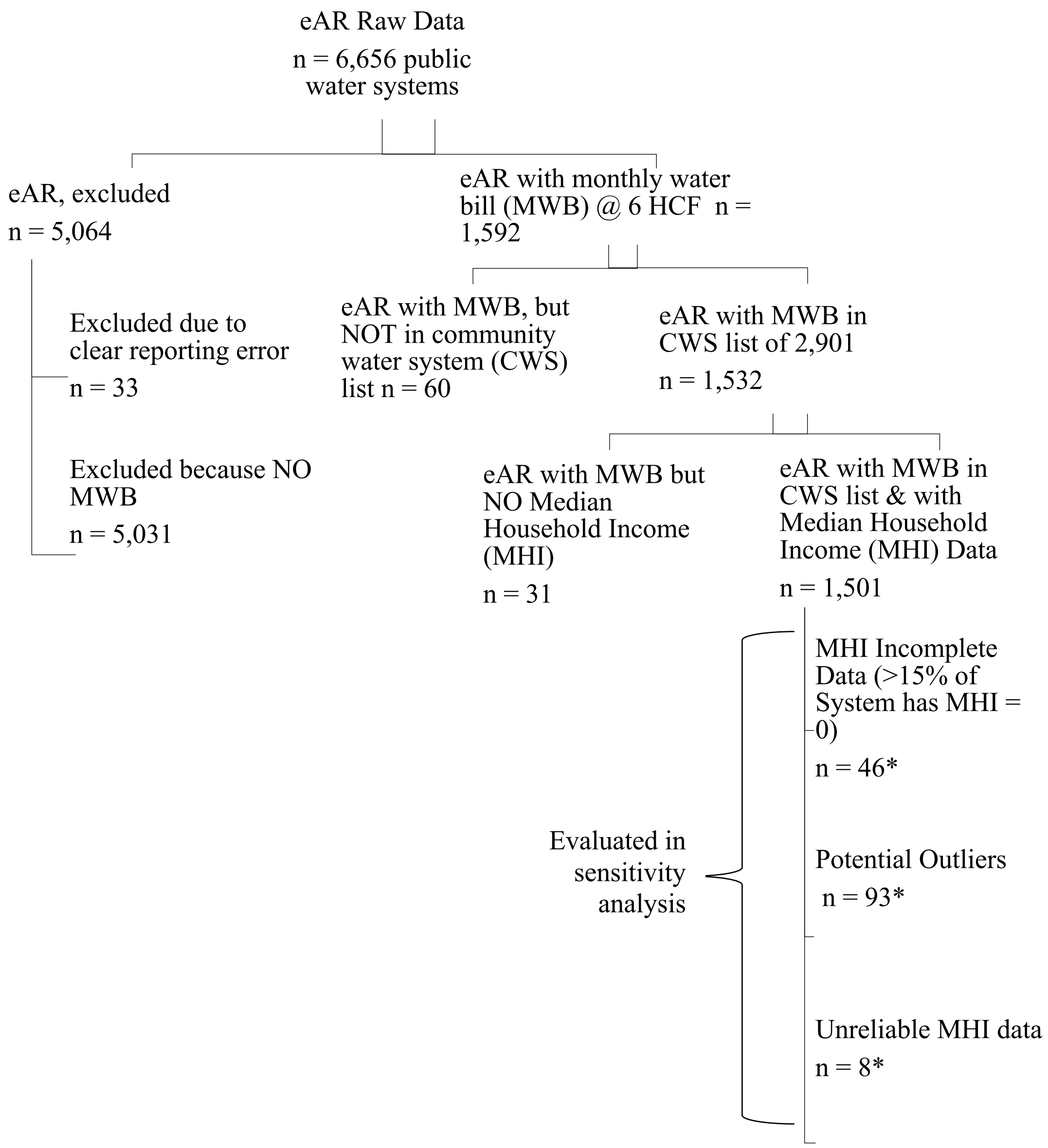
Systems were removed from the analysis of affordability ratios by system size in a sensitivity analysis if they met the following criteria:

1) Water system was above or below the upper and lower fences of our outlier assessment (See B3 below);
2) Water system had over 15% of a system's aerially-weighted households had no available income data (See B4 below)
3) Water system was within one block group and the Median Household Income estimate was considered unreliable in an analysis of Census sampling error (See B4 below).

Systems were removed from the study of household poverty indices by system size in a sensitivity analysis if they met the following criteria:

1) Water system was within one block group and Households within Income bracket data from Census was considered unreliable in an analysis of Census sampling error (See B4 below);

Figure B.1 demonstrates the results of this data cleaning, exclusion, and processing with counts of water systems excluded from raw data.



*Three systems overlap between outlier systems and MHI incomplete data.

Figure B.1 Data cleaning for water bills

Table B.1 Data sources used to create affordability measures and household poverty indices.

Data Type	Data Source	Available:	Manipulation to Match to Water System Boundaries	Excluded in a sensitivity analysis if:	Potential Sources of Error
Water Bills	State Water Resources Control Board electronic Annual Reports (eAR) 2015	https://drinc.ca.gov/drinc/DWPRepository.aspx Updated data retrieved from Division of Drinking Water.	None	Water bill >= $177.32 or <= $13.91 (See B3)	Reporting error or non-response
Median Household Income	ACS Census 2011-2015 5-Year Estimates (Table B19013)	American Fact Finder: https://factfinder.census.gov/faces/tableservices/jsf/pages/productview.xhtml?src=bkmk	Areal-household weighting using water system and block group boundaries (See B2)	Systems that have 15% or more of block groups with missing MHI data (see B4) System is within one block group and has unreliable Census estimate (see B4)	Estimate for water system is not 'true' median but an average of underlying median incomes Assumption of geographic homogeneity in block groups (error increases for more rural geographies) Census data reliability
California Poverty Income Threshold	PPIC California Poverty by County (2015)	PPIC:https://www.ppic.org/map/california-poverty-by-county-and-legislative-district/ County poverty thresholds weighted by number of renters and owners in 2015 retrieved from PPIC.	Water system assigned county poverty income and deep poverty income based on county	None	Of the 58 counties, 38 counties have unique thresholds and the remaining 20 are in 3 groups due to Census suppression criteria (Bohn *et al* 2013a)
Households by Income Bracket & Total Households	ACS Census 2011-2015 5-Year Estimates (Table B19001)	American Fact Finder: https://factfinder.census.gov/faces/tableservices/jsf/pages/productview.xhtml?src=bkmk	Areal-household weighting using water system and block group boundaries (See B2)	Water systems with more than one estimate (for Total Households data) or more than 20% of estimates (for Households by Income Bracket) have unreliable Census estimates (see B4)	Assumption of geographic homogeneity in Census Tracts (error increases for more rural geographies) Census data reliability (i.e. sampling error)

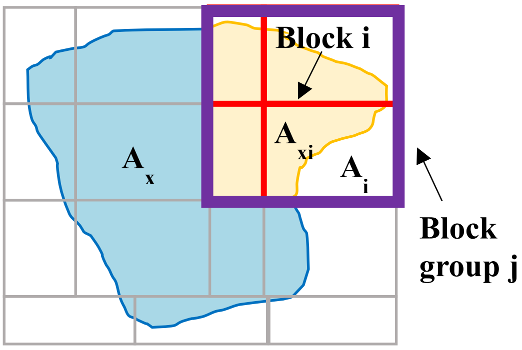
B2 Areal-Household Weighting Methodology

Given a water system boundary that spans several census blocks and census block groups, the following approach was used to attribute census data used in the study from the American Community Survey to water system boundaries. Below, the methodology is demonstrated for use with median household income data, though the same approach applies to other census data in the study (number of households and number of households within income levels).

Intersection of populated blocks with water system boundaries was conducted using ArcGIS 3.0 at OEHHA (see description below). Block level populated household data and shapefiles were obtained from the 2010 Census (Available for download: https://www.census.gov/geo/maps-data/data/tiger-line.html) and overlaid with water system boundaries from the California Environmental Health Tracking Tool website (Water Systems Geographic Reporting Tool available for download: http://cehtp.org/water/map-viewer). Water system boundaries were adjusted according to methods published for CalEnviroScreen 3.0 (OEHHA 2017b).

Let:

Block = i
Block Group= j
Block within a Block Group = ij
Populated household = HH
Area of water system = A_x
Area of block = A_i
Area of block in water system x = A_{xi}



In order to calculate the median household income (MHI) of a given water system (x):

The number of households in block i contained within a block group j contributing to a water system x's area is determined by:

$$HH_{xij} = HH_{ij} \times A_{xi}$$

Where, the A_{xi} is the proportion of block i in the area of the system (A_x):

$$A_{xi} = w_x A_i$$

Where w_x is the aerially determined weight (or the areal difference as a percentage from ArcGIS). The sum of households from blocks i is thus equal to the total households in block group j (BG_j) that intersect with the water system:

$$\sum_{i=1}^{i=n} HH_{xij} = HH_{xj}$$

HH_{xj} is aerially apportioned sum of households associated with block group j in system x. Note: Summing across all block group j would result in an estimate of the total number of households for water system x (HH_x):

$$\sum_{j=1}^{j=n} HH_{xj} = HH_x$$

To obtain the median household income of water system x, we first multiply the MHI associated with block group j by the adjusted household sum for block group j (HH_{xj}) and sum over all block groups associated with system x. Then we divide this sum by the total number of households for the system x:

$$MHI_x = \frac{\sum_{j=1}^{j=n} (MHI_j \times HH_{xj})}{\sum_{j=1}^{j=n} HH_x}$$

For systems flagged as potential unreliable due to missing data in block groups (See B4 below), water system estimates of MHI were calculated excluding the adjusted household sum (HH_{xj}) contributions from block groups with missing data. To calculate the number of households within each income bracket within a water system, we applied the same methodology. To estimate social demographic data used in the study of missing data, we applied the same methodology but used population counts rather than household counts.

B3 Sensitivity Analysis– Water Bills

Given the extreme range of reported values in the eAR survey for water bills across various volumes (i.e. from $3.06 to $466 per month for 6HCF), we conducted a sensitivity analysis, analyzing the results of the study with and without systems with potentially extreme water bill values. The complex nature of water system ownership and heterogeneity among rate structures in California makes a qualitative prior for thresholds challenging to determine. Therefore, we used cut-off points determined from adjusted box plots that account for distribution skew (Hubert and Vandervieren 2008). The upper and lower fences determined by this method served as benchmarks to explore very high and very low water bill values given no prior systematic evaluation of eAR survey data at a fixed volume. It is important to note that the water systems falling above or below the threshold set by the Hubert & Vandervieren (2008) method are statistical outliers, not necessarily real outliers. This results in a conservative approach toward assessing sensitivity in the eAR data.

Of the 1,532 community water systems with useable water bill data (regardless of available income data), 29 systems fell above the upper fence of $181.78, and 69 system fell below the lower fence of $14.79, for a total of 98 potential outlier systems (Table B2).

Table B.2 Identification of upper and lower thresholds used to exclude outliers in sensitivity analysis.

Metrics	Results*
Q1	$29.29
Median of Dataset	$41.56
Q3	$61.60
Interquartile Range (IQR)	$32.32
Medcouple (MC) [+]	0.3
Lower Fence (threshold) = Q1 – [1.5 × exp(-4 × MC) × IQR]	$13.91
Number of systems below lower fence	**69**
Upper Fence (threshold) = Q3 + [1.5 × exp(3 × MC) × IQR]	$177.32
Number of systems above upper fence	**29**

*All calculations were conducted using adjboxStats in the robustbase package of R 3.5.1 (R Core Team 2018)

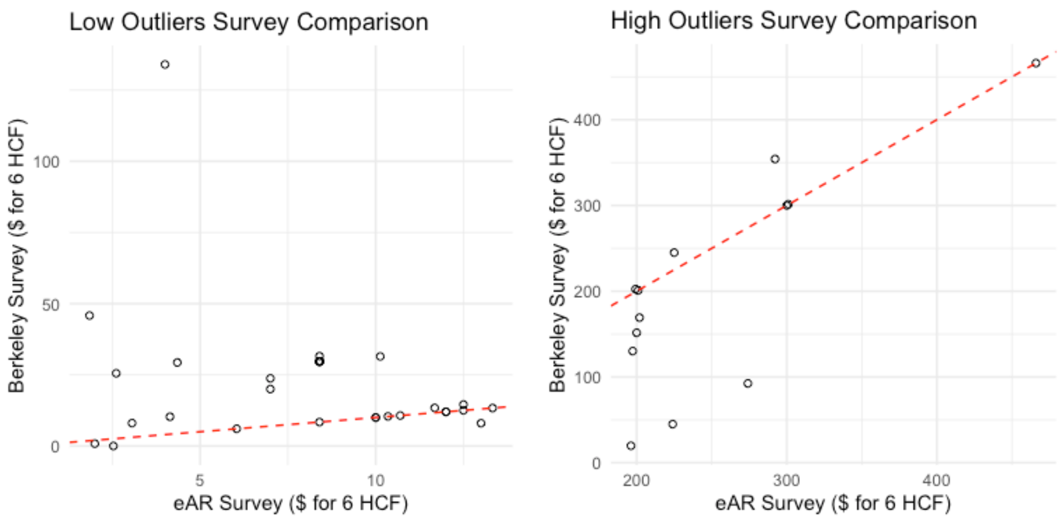
[+]The medcouple is the median of an array calculated using the kernel function as reported in the adjusted box plot method. A positive value (MC > 0) reflects a right-skewed distribution.

Systems above and below the thresholds were cross-listed with a survey of extremely high and low water bills that I conducted in 2015 as part of the Office of Environmental Health Hazard Assessment (OEHHA) Human Right to Water project. At the time, we investigated the reliability of extreme water bills based on upper and lower fences determined given bills for 12 hundred cubic feet (HCF) of water. We asked water systems to answer the same question posted on the Water Board’s electronic annual report survey about rate data and to estimate water bills at 6, 12, and 24 hundred cubic feet. Systems were contacted three times by phone and email before being labeled ‘unreached.’ While we also collected data for 6 HCF water bills for very low and very high water bills, the survey is not directly representative of the systems flagged in this outlier assessment because the original sample was drawn from the 12 HCF water bill data. However, of the 98 systems with affordability data flagged in this study's outlier assessment (as having water bills over $177.32 and below $13.91), 86 were contacted as part of the 12 HCF survey. Of these 86 systems, 37 (43%) responded to the survey.

Figure B.2 compares the water system response to the Berkeley survey question (y-axis) about water bills for 6 HCF versus the eAR survey question (x-axis) for low-outliers and high outliers, respectively. As expected, many systems reporting very low water bills reported higher water bills for 6HCF in the phone survey. The reverse held true for systems responding about potentially very high water bills. Nevertheless, several systems with very low and very high water bills were in fact accurate. These results supported our choice to exclude systems with very low or very high water bills in a sensitivity analysis to determine the effect on our outcomes in the affordability ratio study.

Figure B.2. Results of water system survey to investigate very low and very high water bills.

Original sample design included 86 of the 98 water system flagged as potential outliers in the present study. Red dashed line represents the line of equality between the Berkeley and eAR survey.



B4 Sensitivity Analysis – Missing or Incomplete Data

Median household income (MHI) data were downloaded from the American Community Survey (ACS) 5-Year Table B19013 at the block group scale. Of the 1,532 water systems with water bill data, 31 systems had block groups with no MHI data. Of the 1,501 water systems with water bill, 342 systems contained or overlapped with block groups that had missing MHI data. Of these 342 systems, 46 systems had missing MHI data for more than 15% of the households, as determined by the aerially weighted household contribution of the block group to the water system. These systems were flagged as potentially unreliable for having high amounts of missing data. No data was missing or incomplete for the total number of households or households across income brackets (ACS 5-Year Table B19001).

Census Income Data Reliability

While the ACS already controls for error before publishing results, they provide quantitative information on sample error for their estimates. This is effectively a measure of estimate imprecision. The ACS provides margin of errors (MOE) at 90% confidence levels by quantifying the variance and standard errors of the estimates resulting from the sampling approach using a successive differences replication (SDR) variance estimation methodology (U.S. Census Bureau 2015, 2014). From the variance, ACS calculates the standard error (square root of the variance) and the margin of error at a 90% confidence level:

$$Standard\ Error = \sqrt{Variance}$$

$$Margin\ of\ Error_{90\%\ Confidence\ Interval} = Standard\ Error \times 1.645$$

Given the census provided margin of error estimates, we can back-calculate standard errors to estimate coefficients of variation. The coefficient of variation (COV), equivalent to the relative standard error, measures the ratio between an estimate's standard error and the estimate itself:

$$Coefficient\ of\ Variation = \frac{Standard\ Error}{Estimate} \times 100$$

Coefficients of variation are then used to determine 'reliability' of data points.

Reliability criteria using coefficients of variation

Determining what constitutes a reliable data point based on COVs is not a clear cut decision. Furthermore, for income estimates at the water system scale, we use an areal-household weighting approach to arrive a weighted-average estimate for each water system. To our knowledge, no precedent exists for calculating new MOEs at aggregate geographies that are not simply additive or multiplicative changes within census boundaries. The latest research on census data geography aggregation for error minimization employs a sophisticated algorithm to improve aggregation techniques, but even here the authors work within given census geographies (Spielman and Folch 2015). As such, we develop criteria to flag potentially unreliable census estimates for systems falling within one block group.

We use three sets of estimates: median household income, households by income bracket (16 brackets), and total number of households. Median household income data is used for the affordability ratio at the median household income level (AR_{MHI}). Total households and the number of households in each income bracket are used for creating household indices of poverty in the study (HH_{CP} and HH_{DP}). We use the following exclusion criteria to evaluate census data, as outlined in CalEnviroScreen 3.0 (OEHHA 2017a) for census tracts; in this case, we use block groups:

a) Coefficient of variation greater than 50 (meaning the standard error was less than half of the estimate) and,
b) Standard error was greater than the mean standard error of all California census block groups estimates for the data of interest.

In cases where coefficient of variation is incalculable (e.g. an estimate is 0), we can only look at whether the standard error (MOE/z_{90}) is less than the average, per the second half of the inclusion criteria. We assume that if the standard error is less than the mean of all block groups, the estimate is reliable by the criteria which we can measure.

The household poverty indices (e.g. HH_{CP} and HH_{DP}) estimated for each system are created by summing the number of households below a specified income level; linear interpolation is used between estimates if the income level falls within the census brackets (e.g. between \$20,000-\$25,000). As such, the estimated household index is only impacted by an unreliable estimate if the household index encompasses an unreliable estimate. The unreliable estimate may not impact the interpolation at all. We thus chose a broader criteria and excluded systems in the sensitivity analysis if more than 20% of the estimates were unreliable by our criteria.

RESULTS OF RELIABILITY STUDY FOR SENSITIVITY ANALYSIS

Coefficients of variation for census estimates for systems within one block group

Of the 1,501 systems with water bill and income data, 505 systems fall within one block group. Of the 505 systems with one block group, 429 of them (85%) have fewer than 200 connections. Of systems with only one block group (505 systems), there are 9,090 total estimates to evaluate (18 estimates for each system).

Total households and Median household income estimates. There are no estimates in the total household data that meet the unreliability criteria. Of the 505 systems with data to evaluate reliability, there is one median household income estimate with no margin of error (and thus no coefficient of variation could be calculated). Of the 504 water systems with estimates for median household income, 8 systems had unreliable estimates.

Households by income bracket. Of the 505 systems, 248 systems had 20% of their more than 2 unreliable estimates among the 16 households by income bracket estimates. Of these 248, 5 systems overlapped with the 8 systems found to have unreliable median household income estimates.

Final list systems to exclude in a sensitivity analysis. Of the 1,501 systems with income and water bill data, 8 systems were excluded from the affordability assessment in a sensitivity analysis. Of the 1,501 systems for which we estimated household poverty indices by system size, 227 systems were excluded in a sensitivity analysis.

B5 Sensitivity Analysis Results

Tables B3 and B4 show crude means, standard deviations, adjusted means, and 95% confidence intervals for affordability ratios at county poverty level and deep poverty level, respectively. As the overall result trends for affordability ratios for households earning median income levels did not change across the sensitivity analyses, these results are not shown. The tables demonstrate results of adjusted mean estimates and post-hoc tests for the complete case sample (n = 1,501), the sample less water systems identified as potentially unreliable census data (n=1,447, described in B4), and the sample less water systems identified in the potential water bill outliers assessment (n=1,408, described in B3). Post-hoc tests were conducted after generalized linear models showed a significant difference across affordability ratios for the main effect of interest (water system size), as indicated by F-tests shown in Table 3.4.

As is clear in both tables, removing water systems with unreliable census data and potential water bill outliers results in more differentiation among mean affordability ratios across water system size categories. However, the overall trend–that ratios increase (i.e. become more unaffordable) as system size decreases–holds true across the complete sample and the sensitivity analyses.

Table B.3 Sensitivity analysis for affordability ratios for households earning county poverty level (AR_{CP})‡.

	Affordability Ratio for Households at or Below County Poverty Level HHC_P (%)					
	Complete case (n=1,501)		Complete case less unreliable census (n=1,447)		Complete case less potential outliers (n=1,408)	
Water System Size (pop.)	**Crude means (SD)**	**Adjusted means (95% CI)**	**Crude means (SD)**	**Adjusted means (95% CI)**	**Crude means (SD)**	**Adjusted means (95% CI)**
25-500	2.8 ± 2.2^a	2.2 (2.0, 2.3)	2.8 ± 2.2^a	2.2 (2.1, 2.3)	2.6 ± 1.4^a	2.2 (2.1, 2.3)
501-3,300	2.3 ± 1.6^a	1.9 (1.8, 2.1)	2.2 ± 1.6^b	1.9 (1.8, 2.0)	2.2 ± 1.3^b	1.9 (1.8, 2.1)
3,301-10,000	1.7 ± 0.9^b	1.6 (1.4, 1.7)	1.7 ± 0.7^c	1.6 (1.4, 1.7)	1.8 ± 0.8^c	1.7 (1.5, 1.8)
10,000+	1.5 ± 0.7^b	1.3 (1.3, 1.5)	1.5 ± 0.7^c	1.4 (1.3, 1.5)	1.5 ± 0.7^d	1.5 (1.4, 1.5)

‡ Results are rounded to the tenth of a decimal for percentages. For adjusted means, all data were log transformed for statistical tests and back-transformed for the table. For each measure shown, means that share the same letter column-wise are not significantly different from one another based on Tukey's Honest Difference post-hoc tests on the generalized linear model of affordability ratios including measured confounders. Post-hoc letters were calculated using *multcomp* package in R (Version 3.5.1; R Development Core Team) and ordered to start comparisons with the highest mean value.

Table B.4 Sensitivity analysis for affordability ratios for households earning county deep poverty level (AR_{DP})‡.

	Affordability Ratio for Households at or Below Deep Poverty Level HH_{DP} (%)					
	Complete case (n=1,501)		Complete case less unreliable census (n=1,447)		Complete case less potential outliers (n=1,408)	
Water System Size (pop.)	**Crude means (SD)**	**Adjusted means (95% CI)**	**Crude means (SD)**	**Adjusted means (95% CI)**	**Crude means (SD)**	**Adjusted means (95% CI)**
25-500	5.6 ± 4.3[a]	4.3 (4.1, 4.5)	5.6 ± 4.4[a]	4.3 (4.1, 4.5)	5.2 ± 2.8[a]	4.4 (4.2, 4.6)
501-3,300	4.5 ± 3.2[a]	3.8 (3.6, 4.1)	4.5 ± 3.2[b]	3.8 (3.6, 4.1)	4.4 ± 2.5[b]	3.9 (3.7, 4.1)
3,301-10,000	3.4 ± 1.7[b]	3.1 (2.8, 3.4)	3.4 ± 1.6[c]	3.1 (2.8, 3.4)	3.5 ± 1.7[c]	3.3 (3.1, 3.6)
10,000+	2.9 ± 1.4[b]	2.7 (2.5, 2.9)	2.9 ± 1.4[c]	2.7 (2.5, 2.9)	3.1 ± 1.3[d]	2.9 (2.7, 3.1)

‡ Results are rounded to the tenth of a decimal for percentages. For adjusted means, all data were log transformed for statistical tests and back-transformed for the table. For each measure shown, means that share the same letter column-wise are not significantly different from one another based on Tukey's Honest Difference post-hoc tests on the generalized linear model of affordability ratios including measured confounders. Post-hoc letters were calculated using *multcomp* package in R (Version 3.5.1; R Development Core Team) and ordered to start comparisons with the highest mean value.

Table B5 shows results of sensitivity analysis for households earning at or below county poverty levels across the sample (n=1,501), the same less unreliable income data (n=1,274), the full water system list with water system boundaries to estimate poverty levels (n=2,882), and the full water system list with water system boundaries less unreliable income data (n=2663). As can be seen from the results, removing unreliable data reduces differences across system size categories in household poverty indices for systems in the sample, but not for systems in the full water system list. Trends did not change for households earning at or below deep poverty levels and thus the results are not shown.

Table B.5. Sensitivity analysis for household poverty index (HH_{CP}) ‡

	Households at or below County poverty level HH_{CP} (%)			
System Size (People in System)	**Affordability Sample (n=1501)**	**Affordability sample less unreliable (n=1,274)**	**Water Systems with Boundaries (n=2882)**	**Water Systems with Boundaries less unreliable (n=2,663)**
Very small (<500)	22 ± 13[c]	21 ± 12[b]	24 ± 14[b]	24 ± 13[b]
Small (501-3,300)	28 ± 14[a]	27 ± 14[a]	28 ± 14[a]	27 ± 14[a]
Medium (3,301-10,000)	25 ± 11[a,b]	26 ± 11[a]	25 ± 11[a,b]	25 ± 11[a,b]
Large (10,001+)	24 ± 8[b]	24 ± 8[a]	24 ± 9[b]	24 ± 9[b]

‡ Results are rounded to the nearest integer. All data were square-root transformed for Welch's One-Way ANOVA and post-hoc statistical comparison tests and back-transformed

for the table. For each measure shown, means that share the same letter column-wise are not significantly different from one another based on Games-Howell post-hoc tests for unequal variances. Post-hoc letters were calculated using *userfriendlyscience* and *multcomp* packages in R (Version 3.5.1; R Development Core Team) and ordered to start comparisons with the highest mean value.

B6 Bias Assessment and Measured Confounding

Table B6 provides a simple bias assessment to convey the impact of missing data on the overall study. The columns in white reflect the total list of community water systems and the list of systems in the final study. The columns in the gray reflect all missing systems (for any reason, including no data or inadequate data) and systems that were excluded in a sensitivity analysis based on census unreliability (B4), and based on the outlier assessment (B3). As described in the main chapter, the system list was divided into system size categories and within each size category, measured confounders were analyzed for marginal effects on whether or not a system had missing data. This section describes the bias assessment presented in Table B6 and expands on findings from the study of measured confounding from missingness (Table B7).

System size

System size categories were: Very small (<500 people); Small (501-3,300 people); Intermediate (3,301-10,000 people); Large (10,000+ people). Overall, very small systems are under-represented in the study. We see the effects of this bias in the overall list of systems with missing data (n=1,400)–a disproportionate number of smaller systems do not have data. Very small systems make up about 62% of the full community water system list but 44% of the sample list. The distribution of system characteristics by size is relatively similar between the full community water system list and the systems excluded in a sensitivity analysis due to the water bill outlier criteria, with a slight bias toward excluding disproportionately more medium and large systems in the sensitivity assessment.

Income status

Per California's Water Code definition of disadvantaged community (DAC) status, a system is considered DAC if its median household income is at or below 80% of the state's median, and a severely disadvantaged community (SDAC) if its MHI is at or below 60% of that for the state (Cal. Wat. Code §79505.5 and §13476). For the year of our analysis (2015), MHI across California water systems was $61,818, making SDACs those communities with median incomes below $38,700, and DACs those communities with incomes below $51,600 but above $38,700.

Table B6 shows that the final study list (n=1,501) has a slightly lower percentage of systems (13.5%) characterized as severely disadvantaged (systems with a median household income less than 60% the state of California's MHI for 2015) than the overall community water system list (nearly 18.5%).

Ownership/Governance

There are five ownership categories associated with water systems, but we collapsed these to public versus privately owned systems due to low counts in some ownership category types (Table B6). The final study list (n=1,501) has a higher proportion of public systems relative to private systems (Table B6).

Region

There are eight hydrologic regions that the State Water Board and OEHHA use to designate water system region: Northern California, the Bay Area, the Eastern Sierras, Northern Sierras, San Joaquin Valley, Central Coast, Los Angeles/Southern

California, and Inland Empire/Imperial Valley. Overall, systems in Northern California, Northern Sierra, and San Joaquin Valley are slight underrepresented relative to other regions in the final study list (Table B6). However, systems from Northern California are more likely to be missing only among small systems (Table B7).

Social-demographic data

There is not evidence to suggest that communities that are poorer or wealthier, or those with higher percentages of people identifying with non-White race/ethnicities, would have a higher likelihood of water system reporting data. Nonetheless, significant disparities by race/ethnicity have been identified in water system violations, and therefore we investigated whether missing data is also correlated with poverty and/or race/ethnicity. Estimates of percentage of households identifying as all non-White race/ethnicity categories in the American Community Survey were summed to estimate Percent People of Color (POC) for each block group, following the approach outlined in Appendix B2. % POC was aerially assigned to water system boundaries and population weighted to estimate a % POC by water system. Similarly, the % of households renting and % of households under two times the federal poverty level were aerially and household-weighted to estimate water system level estimates of % Renters and % Under 2X Poverty within each water system.

For % POC, the difference in means between the full system list (n=2,882 with estimates) and the final study list (n=1,501) was significant in a two-way Mann-Whitney U ranked sum difference test (p = 0.03). However, absolute means and standard deviations between systems with and without affordability data were not substantial (Table B6). Systems without affordability data have a slightly lower estimated % POC relative to systems with affordability data. Mann-Whitney U tests between systems with and without data are non-significant for % Renters or % Under Twice the Federal Poverty Level.

For very small systems and intermediate systems, the odds of a system having missing data increased marginally for each unit increase in % POC (OR = 1.01, $p < 0.001$; Table B7). For very small, intermediate, and large systems, the odds of a system having missing data increased marginally for each unit increase in % Renters (OR = 1.01, 1.02, and 1.02 respectively; Table B7). For very small systems, the odds of a system having missing data increased marginally for each unit increase in % Under 2X Poverty (OR = 1.01, $p < 0.001$; Table B7). Social-demographic data does not appear to have a substantial marginal effect on whether or not a system is missing affordability data when stratified by system size. At the same time, though significant differences exist in the distribution of This is somewhat consistent with the fact that a small portion of systems with missing data are missing income-based estimates; rather most missing data is non-reported water bills.

Table B.6 Assessment of bias for systems with missing data for affordability assessment.

	List of all community water systems		**List of systems in final study list**		**Systems in community water system list with missing affordability data**		**Systems in community water system list that did not report water bills**	
	N = 2,901	***% of Systems***	***N = 1,501***	***% of Systems***	***N=1,400***	***% of Systems***	***N = 1,369***	***% of Systems***
<500 people	1,812	62.5	661	44.0	1,151	82.2	1,121	81.9
501-3,300 people	447	15.4	304	20.3	143	10.2	142	10.4
3,301-10,000 people	224	7.7	166	11.1	58	1.1	58	4.2
10,000+ people	418	14.4	370	24.6	48	3.4	48	3.5
	N = 2,901	***% of Systems***	***N = 1,501***	***% of Systems***	***N=1,400***	***% of Systems***	***N = 1,369***	***% of Systems***
MHI < 60% state's MHI	538	18.5	203	13.5	335	23.9	304	22.2
MHI > 60% state MHI but < 80% state's MHI	625	21.5	306	20.4	319	22.8	319	23.3
MHI > 80% state's MHI	1,719	59.3	992	66.1	727	51.9	727	53.1
Unknown	19	0.7	0	0	19	1.4	0	1.4
	N = 2,901	***% of Systems***	***N = 1,501***	***% of Systems***	***N=1,400***	***% of Systems***	***N = 1,369***	***% of Systems***
Publicly owned System	1,070	36.9	713	47.5	357	25.5	341	24.9
Privately owned System	1,831	63.1	788	52.5	1,043	74.5	1,028	75.1
	N = 2,901	***% of Systems***	***N = 1,501***	***% of Systems***	***N=1,400***	***% of Systems***	***N = 1,369***	***% of Systems***
Northern California	479	16.5	219	14.6	260	18.5	255	18.6
Bay Area	362	12.5	198	13.2	164	11.7	163	11.9
Eastern Sierras	199	3.6	96	6.4	103	7.3	100	7.3
Northern Sierras	170	5.9	74	4.9	96	6.6	92	6.7
San Joaquin Valley	654	22.5	282	18.8	372	26.6	361	26.4
Central Coast	374	12.9	203	13.5	171	12.2	167	12.2
Los Angeles/So.Cal	389	13.4	275	18.3	114	8.1	113	8.3
Inland Empire	274	9.4	154	10.3	120	8.6	118	8.6
	Mean ± SD		**Mean ± SD**		**Mean ± SD**		**Mean ± SD**	
	N = 2,901		**N = 1,501**		**N = 1,400**		**N = 1,369**	
% People of Color	40.7 ± 26.5		42.4 ± 26.5		38.9 ± 26.4		39.2 ± 26.4	
% Renters	33.8 ± 18.0		34 ± 16.4		33.7 ± 19.7		33.9 ± 19.6	
% Under 2X Poverty	35.9 ± 18.4		34.6 ± 17.7		37.2 ± 19.2		37.2 ± 19.2	

Table B.7. Results of logarithmic predictions of missing data within size categories, by potential confounders of missing affordability data.

Each row by column cell reflects a unique binomial regression equation results. Significant results (alpha < 0.05) presented.

System Size (People Served)	n Missing / n Not Missing Data	Region *reference = Northern California*		Primary Source *reference = Surface water (SW)*		System Ownership *reference = Public*		% Renters	% Under 2x Poverty	% People of Color
<500	1,151 / 661	Nor Cal☆ Bay Area☆ C. Coast☆	OR=1.97*** OR=0.61** OR=0.59***	GW☆	OR=1.44*	Public☆ Private☆	OR=1.35** OR=1.37*	☆OR=1.01***	☆OR=1.01***	☆OR=1.01**
501-3,300	143 / 304	Nor Cal	OR=0.43***	SW☆ GW☆	OR=0.33*** OR=1.63*	Public	OR=0.43***	--	--	--
3,301-10,000	58 / 166	Nor Cal Nor Sierra SJV	OR=0.15*** OR=4.00* OR=3.61*	SW	OR=0.29***	Public Private	OR=0.41*** OR=0.43*	☆OR=1.02**	--	☆OR=1.01*
10,000+	48 / 370	Nor Cal	OR=0.13**	☆GW ☆SW	OR=0.09*** OR=2.57**	☆Public ☆Private	OR=0.16*** OR=0.14**	☆OR=1.02*	--	--

* $p < 0.05$; ** $p < 0.01$; *** $p < 0.001$

OR = odds ratio, or, the odds of a system having missing data. OR > 1 and significant at alpha <0.05 means a significant *increase* in odds of a system having missing data. OR <1 and significant at alpha < 0.05 means a significant *decrease* in the odds of a system having missing data. **☆ = Mann Whitney's U test** (for continuous variables) or **Chi-squared (χ^2) test** (for categorical variables) was significant at alpha <0.05. Significant Mann Whitney's U test indicates a location shift greater or less than zero between the rank sums of systems missing data and those not missing data. Significant χ^2 test indicates a rejection of the null that systems with missing data are as likely to have the same frequency of systems in each region/ownership/primary source distribution as those systems without missing data. For Intermediate and Large systems, low counts of systems by region meant that χ^2 tests were not stable and therefore we conducted Fisher's exact tests using simulated p-values. In neither system size category were the count distributions significantly different between systems with and without missing data.

B7 AFFORDABILITY RATIOS BY COMMON AFFORDABILITY THRESHOLDS

Table B8 summarizes common affordability thresholds and references, with results from study compared to these thresholds where applicable.

Table B.8. Common affordability thresholds and results for AR_{MHI}, AR_{CP}, AR_{DP}, where potentially comparable.

Affordability Ratio Threshold	Water Cost Included	Reference	% of Systems in Study Exceeding Threshold
1.5% of MHI (specifically in disadvantaged communities)	Drinking water services	(State Water Resources Control Board 2018, Pierce and McCann 2015)	AR_{MHI}: 18.7% (n = 281) AR_{MHI}: 11.4% (n = 172 disadvantaged systems)
2% of MHI	Wastewater services	(US EPA 1997)	N/A
2% of MHI	Drinking water services	(Hanak *et al* 2014, Fong 2012, Christian-Smith *et al* 2013)	AR_{MHI}: 11.2% (n = 168)
2.5% of MHI	Drinking water services – for compliance purposes	(EPA Science Advisory Board 2002, US EPA 1998a)	AR_{MHI}: 6.3% (n = 95)
3% of income (often referenced for drinking water alone and/ or for disposable incomes)	Drinking water & wastewater services	(2010, Barraqué and Montginoul 2015, Fitch and Price 2002, United Nations Development Program 2006)	AR_{CP}: 19% (n = 285) AR_{DP}: 62% (n = 937)
4.5% of MHI	Drinking water & wastewater services	(Mack and Wrase 2017)	N/A
5% of MHI	Drinking water & wastewater services	(Banerjee and Morella 2011, Villumsen and Jensen 2014)	N/A
5% of discretionary income (for 20th income percentile)	Drinking water services	(Feinstein 2018, Teodoro 2018, 2019)	AR_{CP}: 5.7% (n = 85) AR_{DP}: 28.7% (n = 431)

APPENDIX C

C1 INDIRECT AND DIRECT EMISSIONS IN CONVERSION FACTORS

In the United States and the UK, the analysis of greenhouse gas emissions associated with fossil fuels over their life cycle has been spearheaded by the transportation sector (Edwards *et al* 2011, National Energy Technology Laboratory (NETL) 2008). Defra/DECC calculate indirect emissions with data from a widely used European study that analyzes the impact of typical fuels used in the transportation sector (Hill *et al* 2013, Edwards *et al* 2011). Defra/DECC data are both more comprehensive and simpler to extract for the purposes of developing the LCA library than the U.S. data.

Indirect emissions calculated in the European context are unlikely to reflect the Mexican case as well as those of direct emissions, because indirect emissions primarily account for the extraction, transportation and refining of fuels. This is likely to be highly variable based on the fuel source geography and fuel distribution routes in Mexico. However, there is one observable trend in the ratio of indirect to direct emissions of fossil fuels that suggests the Defra/DECC data for indirect emissions can be reasonably applied to the current problem given the lack of other data. NETL (2008) found that the direct emissions phase of a fuel (i.e., the use phase) accounts for 79.6% to 83.7% of the life cycle greenhouse gas emissions (National Energy Technology Laboratory (NETL) 2008). Specifically, NETL (2008) found that indirect emissions for gasoline, diesel, and kerosene account for 16.3% to 20% of life cycle emissions generally. Calculating this ratio for the indirect to direct emissions published by Defra/DECC reveals an indirect emissions composition of 16.7% to 20.4% for the same three fuels (Table C1). Inclusion of Natural Gas, LPG, among other fuels, the ratio of indirect to direct emissions spans from 11.1% to 20.4% (Table C1).

Defra/DECC conversion factors are within a reasonable range of other LCA studies and follow a similar trend regarding indirect to direct emissions values. Given a lack of Mexico specific emissions factors, we use DEFRA and DECC emissions factors as an approximation for conversion factors in Mexico.

We do not expect the direct emissions values constructed in the UK case to differ greatly from the Mexican case, but as there is no direct data for comparison with Mexico, several other LCA studies from the United States were compared to the Defra/DECC data for several fuels (Table C1).

Table C.1 Common indirect and direct emissions estimates by source available during year of study.

Fuel	Scenario	Direct Emissions	Indirect Emissions	Total	Units	Source
Gasoline/ Petrol	Low	0.42	2.4	2.88	kg CO2eq/liter	(Roundtable on Sustainable Biofuels 2011)*
	Average	0.56	2.41	2.97	kg CO2eq/liter	(Roundtable on Sustainable Biofuels 2011)*
	High	0.69	2.42	3.05	kg CO2eq/liter	(Roundtable on Sustainable Biofuels 2011)*
	--	0.61	2.39	3	kg CO2eq/liter	(National Energy Technology Laboratory (NETL) 2008)
	--	--	2.33	--	kg CO2eq/liter	(US EPA 2011)
		0.46	2.31	2.78	kg CO2eq/liter	(AEA for DECC and Defra 2012)
Diesel	Low	0.46	2.63	3.15	kg CO2eq/liter	(Roundtable on Sustainable Biofuels 2011)*
	Average	0.61	2.64	3.25	kg CO2eq/liter	(Roundtable on Sustainable Biofuels 2011)*
	High	0.76	2.65	3.34	kg CO2eq/liter	(Roundtable on Sustainable Biofuels 2011)*
	--	0.63	2.62	3.25	kg CO2eq/liter	(National Energy Technology Laboratory (NETL) 2008)
	--	0.56	2.68	3.24	kg CO2eq/liter	(AEA for DECC and Defra 2012)
Kerosene	Low	0.4	2.27	2.72	kg CO2eq/liter	(Roundtable on Sustainable Biofuels 2011)*
	Average	0.53	2.28	2.8	kg CO2eq/liter	(Roundtable on Sustainable Biofuels 2011)*
	High	0.65	2.29	2.88	kg CO2eq/liter	(Roundtable on Sustainable Biofuels 2011)*
	--	0.45	2.26	2.74	kg CO2eq/liter	(National Energy Technology Laboratory (NETL) 2008)
	--	--	2.58	--	kg CO2eq/liter	(US EPA 2011)
	--	0.53	2.55	3.08	kg CO2eq/liter	(AEA for DECC and Defra 2012)

Natural Gas	--	0.27	--	--	kg CO2eq/m3	(Spath and Mann n.d.)
	--	0.3	--	--	kg CO2eq/m3	(Jaramillo *et al* 2007)
	--	0.56	--	--	kg CO2eq/m3	(National Energy Technology Laboratory (NETL) 2011)
	--	0.43	--	--	kg CO2eq/m3	(AEA for DECC and Defra 2012)
	--	--	1.93	--	kg CO2eq/m3	(US EPA 2011)
	--	0.21	2.03	2.24	kg CO2eq/m3	(AEA for DECC and Defra 2012)
Mixed Coal (Electric Power Sector)	--	131.73	2195.46	2327.18	kg CO2eq/tonne	(Jaramillo *et al* 2007)
	--	--	2069	--	kg CO2eq/tonne	(US EPA 2011)
	--	369.3	2258.2	2627.5	kg CO2eq/tonne	(AEA for DECC and Defra 2012)
Mixed Coal (Industrial)	--	--	2332.36	--	kg CO2eq/tonne	(US EPA 2011)
	--	400.5	2183.8	2584.3	kg CO2eq/tonne	(AEA for DECC and Defra 2012)
Liquified Petroleum Gas (LPG)	--	--	1.54	--	kg CO2eq/liter	(US EPA 2011)
	--	0.19	1.53	1.72	kg CO2eq/liter	(AEA for DECC and Defra 2012)
Naptha	--	--	2.25	--	kg CO2eq/liter	(US EPA 2011)
	--	0.3	2.13	2.43	kg CO2eq/liter	(AEA for DECC and Defra 2012)
Distilled Fuel Oil	--	--	2.71	--	kg CO2eq/liter	(US EPA 2011)
Fuel oil	--	0.6	3.18	3.78	kg CO2eq/liter	(AEA for DECC and Defra 2012)

Notes on Table

a) Conversions of original data from Defra/DECC GHG equivalence done using "Fuel Properties" (Appendix 11 in Defra/DECC calculator).

b) Range values of low, average, and high represent the uncertainty values reported.

c) CO_{2eq} emissions from EPA estimated using emission factors provided in EPA's Emission Factors Hub, November 2011 (http://www.epa.gov/climateleaders/guidance/ghg-emissions.html).

d) Distillate Fuel Oil reported as "Fuel Oil".

e) In accordance with UNFCCC reporting, both the EPA and the Defra/DECC documents use Global Warming Potentials from the IPCC Second Assessment Report despite the fact that they have been revised since 2007.